EAST WATERWAY OPERABLE UNIT
SUPPLEMENTAL REMEDIAL INVESTIGATION/ FEASIBILITY STUDY
ERA TECHNICAL MEMORANDUM
FINAL

For submittal to:

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Region 10
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Prepared by: Windward environmental LLC

200 West Mercer Street, Suite 401
Seattle, Washington ♦ 98119

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Acronyms

ACRONYM	Definition
AET	apparent effects threshold
AWQC	ambient water quality criteria
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
COI	chemical of interest
COPC	chemical of potential concern
CSL	cleanup screening level
CSM	conceptual site model
CSO	combined sewer overflow
DMMP	Dredged Material Management Program
dw	dry weight
EISR	existing information summary report
EPA	US Environmental Protection Agency
EPC	exposure point concentration
ERA	ecological risk assessment
ESG	Environmental Solutions Group
EW	East Waterway
FS	feasibility study
HPAH	high-molecular-weight polycyclic aromatic hydrocarbon
HQ	hazard quotient
IR	ingestion rate
IRI	index of relative importance
KC	King County
LC50	concentration that is lethal to 50% of an exposed population
LDW	Lower Duwamish Waterway
LOAEL	lowest-observed-adverse-effects level
LPAH	low-molecular-weight polycyclic aromatic hydrocarbon
ML	maximum level
NOAEL	no-observed-adverse-effects level
OC	organic carbon
PAH	polycyclic aromatic hydrocarbon
PCB	polychlorinated biphenyl

ACRONYM	Definition
PDM	post-dredge monitoring
PSAMP	Puget Sound Ambient Monitoring Program
PSEP	Puget Sound Estuary Program
QAPP	quality assurance project plan
RI	remedial investigation
ROC	receptor of concern
RPS	relative penis size
SL	screening level
SMS	Washington State Sediment Management Standards
SQS	sediment quality standards
SRI	supplemental remedial investigation
SVOC	semivolatile organic compound
T-18	Terminal 18
TBT	tributyltin
TCDD	tetrachlorodibenzo-*p*-dioxin
TEF	toxic equivalency factor
TEQ	toxic equivalent
TOC	total organic carbon
TRV	toxicity reference value
UCL	upper confidence limit on the mean
USCG	US Coast Guard
USFWS	US Fish and Wildlife Service
USGS	US Geological Survey
VOC	volatile organic compound
WAC	Washington Administrative Code
WDFW	Washington State Department of Fish and Wildlife
WHO	World Health Organization
Windward	Windward Environmental LLC
WQA	water quality assessment
WQS	water quality standards
WSOU	Waterway Sediment Operable Unit

1 Introduction

This technical memorandum outlines the framework for the baseline ecological risk assessment (ERA) for the East Waterway (EW) Operable Unit supplemental remedial investigation (SRI) and feasibility study (FS). This document describes the methods and approaches based on the US Environmental Protection Agency's (EPA's) ERA guidance for conducting risk assessments under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) (EPA 1992, 1997a, b, 1998). To the extent appropriate, this ERA is consistent with the approach and methods approved by EPA for use in the ERA for Lower Duwamish Waterway (LDW), which is upstream of the EW.

The overall objective of this document is to ensure agreement on the proposed ERA approach prior to conducting the risk assessment. This document presents the process used to identify chemicals of interest (COIs), and subsequently, chemicals of potential concern (COPCs) for each medium and receptor of concern (ROC) that will be addressed in the ERA. This process includes the evaluation and selection of toxicity reference values (TRVs) that will be used to identify COPCs and assess risks. Also provided are the methods for conducting the exposure and effects evaluation and the risk characterization of the ERA, including brief descriptions of key regulatory values, toxicity thresholds, exposure parameters, and exposure assumptions proposed for use in the ERA.

Datasets and rules for data reduction (e.g., calculation of chemical group totals, treatment of reporting limits) that will be applied to the ERA are also identified.

2 Problem Formulation

This section describes the technical approach for the problem formulation of the ERA. The problem formulation section consists of the following components:

- Description of the environmental setting and habitat features of the EW
- Description and selection of the ROCs
- Description of datasets available to support the ERA
- Selection of COPCs
- Refinement of the ecological conceptual site model (CSM)

The CSM for the EW ERA was presented in Anchor and Windward (2008), along with a description of the environmental setting, receptors potentially at risk, and available data. The ROCs selected in the draft CSM report are: benthic invertebrate community (assemblages of small invertebrates that live on or in the sediment), cancrid crabs, juvenile Chinook salmon, English sole, brown rockfish, osprey, pigeon guillemot, river otter, and harbor seal. These receptors reflect the overall management goals for the site that include:

- Limit/reduce exposure of the benthic invertebrate community to sediment contaminants to concentrations below which no adverse effects on the benthic invertebrate community occur.
- Limit/reduce exposure of crabs, fishes, birds and mammals to sediment associated contaminants to concentrations below which no adverse effects on populations occur.
- Limit/reduce exposure of migratory juvenile salmonids to sediment-associated contaminants to concentrations below which no adverse effects on individuals occur.

An overview of the lines of evidence to be used in the risk evaluation, as initially presented in the CSM report, is shown in Table 2-1. These lines of evidence include the measures of exposure and effects for each of the ROCs. This technical memorandum presents a further discussion of available data, which include the new (i.e., 2008) and proposed SRI data, and also describes the COPC selection process that will be used in the problem formulation of the ERA.

Table 2-1. Lines of evidence for risk evaluation for the selected ecological receptors of concern

Receptor of Concern	Line of Evidence: Assessment Endpoint	Line of Evidence: Measure of Exposure	Line of Evidence: Measure of Effect	Method of Evaluation
Benthic Invertebrate Community				
Benthic invertebrate community (infauna/ epifauna)	maintenance of the benthic invertebrate community in EW sediment	chemical concentrations in surface sediment[a]	SMS[b] and toxicity-based regional guidelines (where no standards are available)	Compare measured chemical concentrations in sediment to Washington State SMS or DMMP guidelines.
			site-specific sediment toxicity tests (survival and growth) relative to reference area sediment toxicity tests	Compare 10-day amphipod survival in site sediment to amphipod survival in reference area sediment.
				Compare 48-hr echinoderm embryo or bivalve larvae normal survival in site sediment elutriates with normal embryo/larval survival in reference area sediment.
				Compare 20-day polychaete growth in site sediment with polychaete growth in reference area sediment.
		VOC concentrations in porewater[c]	WQS and AWQC	Compare chemical concentrations in porewater to WQS, AWQC, or literature-based TRVs when no standards/criteria are available.
		PCB, mercury, and TBT concentrations in benthic invertebrate tissue (field-collected)	tissue-residue TRVs based on survival, growth, and reproduction	Compare measured tissue burdens to tissue-residue TRV.
Crabs				
Cancrid crabs	maintenance of crab populations in the EW	concentrations of chemicals in cancrid crab whole-body[d] tissue	tissue-residue TRVs based survival, growth, and reproduction	Compare chemical concentrations measured in tissue to tissue-residue-based TRVs for crabs or other decapods.

Receptor of Concern	Line of Evidence			Method of Evaluation
	Assessment Endpoint	Measure of Exposure	Measure of Effect	
Fish				
Juvenile Chinook salmon	survival and growth of individual juvenile anadromous salmon in the EW	chemical concentrations in juvenile Chinook salmon whole-body tissue	tissue-residue TRVs[e, f] based on survival and growth	Compare chemical concentrations in juvenile Chinook tissue to fish tissue-residue TRVs.
		chemical concentrations in prey (benthic invertebrate) tissue	dietary TRVs[e, g] based on survival and growth	Compare chemical concentrations in juvenile Chinook salmon prey and juvenile Chinook salmon stomach contents to diet-based TRVs for fish.
		chemical concentrations in juvenile Chinook salmon stomach contents		
		chemical concentrations in surface water	WQS, AWQC, and water TRVs based on survival and growth	Compare chemical concentrations in surface water to WQS, AWQC, or other relevant TRVs.
English sole	maintenance of benthivorous and planktivorous fish populations in the EW	chemical concentrations in English sole whole-body tissue	tissue-residue TRVs[f] based on survival, growth, and reproduction	Compare chemical concentrations in English sole tissue to fish tissue-residue TRVs.
		chemical concentrations in prey (benthic invertebrate) tissue and surface sediment	dietary TRVs[g] based on survival, growth, and reproduction	Compare chemical concentrations in English sole prey and incidentally ingested surface sediment collected throughout the EW to diet-based TRVs for fish.
		chemical concentrations in surface water	WQS, AWQC, and other water TRVs based on survival, growth, and reproduction	Compare chemical concentrations in surface water to WQS, AWQC, or other relevant TRVs.

Receptor of Concern	Line of Evidence			Method of Evaluation
	Assessment Endpoint	Measure of Exposure	Measure of Effect	
Brown rockfish	maintenance of upper-trophic-level fish populations in the EW	chemical concentrations in brown rockfish whole-body tissue	tissue-residue TRVs[f] based on survival, growth, and reproduction	Compare chemical concentrations in rockfish tissue to tissue-residue TRVs for fish.
		chemical concentrations in prey tissue (benthic invertebrate, shrimp[h], juvenile Chinook salmon, shiner surfperch) and surface sediment	dietary TRVs[g] based on survival, growth, and reproduction	Compare chemical concentrations in brown rockfish prey and incidentally ingested surface sediment collected throughout the EW to diet-based TRVs for fish.
		chemical concentrations in surface water	WQS, AWQC, and other water TRVs based on survival, growth, and reproduction	Compare chemical concentrations in surface water to WQS, AWQC, or other relevant TRVs.
Wildlife				
Osprey[i]	maintenance of piscivorous bird populations in the EW	chemical concentrations in prey fish tissue and surface water	dietary TRVs based on survival, growth, and reproduction of birds	Compare dietary dose calculated from chemical concentrations in fish, surface water, and incidentally ingested sediment to diet-based TRVs for birds.
Pigeon guillemot	maintenance of piscivorous/ benthivorous bird populations in the EW	chemical concentrations in prey (fish tissue, shrimp, crabs, and mussels), surface sediment, and surface water	dietary TRVs based on survival, growth, and reproduction of birds	Compare dietary dose calculated from chemical concentrations in fish, invertebrates, incidentally ingested surface sediment, and surface water to diet-based TRVs for birds.
River otter	maintenance of piscivorous semi-aquatic mammal populations in the EW	chemical concentrations in prey (fish tissue, clams, crabs, and mussels), surface sediment, and surface water	dietary TRVs based on survival, growth, and reproduction of mammals	Compare dietary dose calculated from chemical concentrations in fish, invertebrates, incidentally ingested surface sediment, and surface water to diet-based TRVs for mammals.
Harbor seal	maintenance of piscivorous marine mammal populations in the EW	chemical concentrations in prey fish tissue, surface sediment, and surface water	dietary TRVs based on survival, growth, and reproduction of mammals	Compare dietary dose calculated from chemical concentrations in fish, incidentally ingested surface sediment, and surface water to diet-based TRVs for mammals.

a Surface sediment is defined as the biologically active zone.

b SMS standards are based on alterations in the benthic community structure, decrease in luminescence of bacteria (assuming toxicity as the primary factor), and direct toxicity to bivalve larvae (measured as combined mortality and abnormal growth) and amphipods (measured as mortality).

[c] Porewater will only be evaluated from intertidal areas adjacent to upland sites with known AWQC/WQS exceedances for VOCs in groundwater or historical site uses that may have resulted in the release of VOCs (if such areas are identified).

[d] Although, edible tissue and the hepatopancreas will be analyzed separately to support the human health assessment, a surrogate whole-body concentration will be derived as a weighted sum of these two tissues and compared to a TRV relevant to the ROC. Chemical concentrations in hepatopancreas tissue are likely to provide a conservative estimate of internal organs in general because the hepatopancreas constitutes the great majority of organ mass and has a relatively high lipid content relative to other organs.

[e] Growth and survival endpoints will be preferentially evaluated for assessing risks to juvenile Chinook salmon because egg and embryo life stages do not occur in the EW and because their exposure in the EW as adults is limited. However, if for a given COPC, a reproduction-based TRV is selected for other fish receptors, the effects data for the salmon TRV will be evaluated to ensure that the TRV is protective of sub-lethal effects for the salmon.

[f] A tissue-residue-based approach, wherein the whole-body tissue chemical concentration in the ROC is compared to a tissue-based TRV, will be used for organic chemicals (except PAHs) and TBT, mercury, and selenium because tissue concentrations account for exposures from all pathways and reflect the concentration at the site of action better than concentrations in exposure media (i.e., diet and water). Chemicals evaluated using the tissue-residue approach will not be evaluated using the dietary LOE. However, because the water effects dataset is often more robust than the tissue residue effects dataset, chemicals evaluated using the tissue-residue approach will also be evaluated using the water line of evidence. Risk estimates from the tissue residue and water LOEs will be reconciled in a weight-of-evidence analysis.

[g] An assessment of exposure through the diet and water will be used for PAHs and dietary metals (arsenic, antimony, cadmium, chromium, cobalt, copper, lead, molybdenum, nickel, silver, thallium, zinc) because fish readily metabolize PAHs and regulate their body burden of metals; thus, tissue concentrations of these chemicals poorly reflect concentrations associated with adverse effects.

[h] Analysis of exposure at a smaller than site-wide scale will be conducted using mussel tissue as a surrogate to represent the shrimp fraction of the diet.

[i] The USGS and USFWS collected osprey eggs from nests near the EW in 2006 and 2007 for chemical analysis (Davis 2007). If these data become available in time to incorporate them into the ERA, they may be compared to TRVs for egg concentrations as an additional line of evidence.

AWQC – ambient water quality criteria
COI – chemical of interest
DMMP – Dredge Material Management Program
EW – East Waterway
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SMS – Washington State Sediment Management Standards
TBT – tributyltin
TRV – toxicity reference value
USGS – US Geological Survey
USFWS – US Fish and Wildlife Service
VOC – volatile organic compound
WQS –water quality standards

2.1 Available and Proposed Datasets

The types of data available for use in the ERA are:

- Surface sediment chemistry data
- Tissue chemistry data for benthic invertebrates (small infauna/epifauna), clams, mussels, shrimp, crabs, English sole, shiner surfperch, brown rockfish, and juvenile Chinook salmon
- Surface water chemistry data
- Site-specific sediment toxicity test results
- Sediment profile images
- Porewater chemistry data

Rules regarding data reduction, derivation of chemical group totals, treatment of undetected chemicals, and total organic carbon (TOC) normalization of sediment chemistry are described in Appendix A (data management). The remainder of this section discusses available data collected from the EW and the data selection process that will be used in the ERA. Final data selected for use will be identified in the ERA report.

2.1.1 Surface sediment chemistry

The following considerations will be made in selecting existing surface sediment data for the ERA dataset:

- **Depth of sample –** Only sediment collected from the uppermost 10 cm will be included.
- **Sampling date –** Only data collected after 1994 will be included.
- **Dredging activities –** Only data collected from locations that were not subsequently dredged will be included.
- **Data quality –** Only data that are considered acceptable based on data validation results will be included (historical datasets were reviewed in the existing information summary report [EISR] (Anchor and Windward 2008).

A substantial number of surface sediment samples have been collected from the EW (Table 2-2).

Table 2-2. Summary of available and proposed surface sediment data for potential use in the EW

Year of Sample Collection	No. of Samples[a]	Analytes[b]	Sample Depth (cm)	No. of Dredged Samples	Event Name	Source
2009	99	SMS, pesticides, dioxins and furans and PCB congeners[c]	0 – 10	0	EW SRI	Windward 2009 (in prep)
2007	3	SMS, dioxins and furans	0 – 10	0	PSAMP sampling	preliminary data
2007	24	DMMP	0 – 10	0	EW – Recontamination Monitoring 2007	Windward (2008a)
2007	7	SMS, pesticides, TBT	0 – 10	0	EW – Slip 27	Windward (2007a)
2006	21	DMMP	0 – 10	0	EW – Recontamination Monitoring 2006	Windward (2007b)
2005	13	SMS	0 – 10	0	USCG (Pier 36-37 slip and Berth Alpha)	Hart Crowser (2005)
2005	53	SMS; DMMP	0 – 10	0	Phase 1A Removal Post-dredge Monitoring	Anchor and Windward (2005)
2001	43[d]	SMS; DMMP	0 – 10	2	EW/Harbor Island Nature and Extent – Phases 1 and 2	Windward(2002a)
2000	13	SMS; DMMP	0 – 10	0	T-18 – post-dredge monitoring	Windward(2001)
1997	3	SMS	0 – 10	3	Pier 36/37 - surface	Tetra Tech (Tetra Tech 1997)
1996	3	SMS	0 – 10	0	Pier 36 - underpier	Tetra Tech (Tetra Tech 1996)
1996	6	SMS	0 – 2	2	King County CSO 96	King County (1996)
1995	7	SMS	0 – 2	2	King County CSO 95	King County (METRO 1995)
1995	12	SMS	0 – 10	9	Harbor Island SRI	EVS (EVS 1996a, b)

[a] The total number of samples analyzed as part of the original investigation within the EW boundary, including samples that were characterized for removal but were subsequently not removed.

[b] SMS analytes include PCBs, SVOCs, metals (arsenic, cadmium, chromium, copper, lead, mercury, silver, and zinc), TOC, and grain size; DMMP analytes include PCBs, pesticides, SVOCs, TBT, metals (antimony, arsenic, cadmium, chromium, copper, lead, mercury, nickel, silver, and zinc), VOCs, TOC, and grain size.

[c] Composite samples were analyzed for PCB congeners and dioxins and furans. Intertidal areas were sampled to create multi-increment samples. Subtidal composite sediment samples were created for 13 areas that cover the entire study site.

[d] Samples were collected from 43 locations. An undisturbed sediment aliquot was first removed from each sample for VOC analysis and given a sample identifier; this was different from the homogenized sample submitted for other chemical analyses.

CSO – combined sewer overflow
DMMP – Dredged Material Management Program
EW – East Waterway
na – not available
SRI – supplemental remedial investigation
SVOC – semivolatile organic compound
T-18 – Terminal 18
TBT – tributyltin

PCB – polychlorinated biphenyl
PSAMP – Puget Sound Ambient Monitoring Program
RI – remedial investigation
SMS – Washington Sediment Management Standards
TOC – total organic carbon
VOC – volatile organic compound
USCG – US Coast Guard

2.1.2 Tissue chemistry

A variety of tissue samples have been collected from the EW (Table 2-3). The 2008 SRI sampling included the collection of English sole, brown rockfish, shiner surfperch, crabs, mussels, shrimp, benthic invertebrates, and clams. Juvenile Chinook salmon (including stomach tissue) will be collected in 2009. The sampling design for the SRI tissue collection events included consideration of the following for data representativeness in the SRI:

- Home range of species relative to the site
- Prey species and preferred prey size of ROCs
- Availability of tissue types
- Age of the data

A data quality review will be conducted, and only those data that are considered acceptable based on data validation results will be included in the ERA. Historical data were reviewed as part of the EISR. Only tissue data collected in 1995 or later will be included in the ERA.

Table 2-3. Summary of available and proposed tissue data for potential use in the EW

Species	Year of Sample Collection	No. of Samples	No of Individuals per Sample	Sample Type	Analytes	Event Name	Source
English sole	2008	11	5	whole body	PCBs (Aroclors), pesticides, SVOCs, metals, inorganic arsenic, butyltins, lipids, dioxins and furans (subset of samples), PCB congeners (subset of samples)	EW-Fish Collection 2008	Windward (2008d)
		11	5	skinless fillet			
	2005	2	5	skinless fillet and remainder[a]	PCBs (Aroclors), mercury, lipids,	EW-Fish Collection 2005	Windward (2005)
	1995	3	6 to 8	skinless fillet	PCBs (Aroclors and subset of congeners), butyltins, mercury, methylmercury, lipids,	EVS 95	Battelle (1996), Frontier GeoSciences (1996)
Brown rockfish	2008	14	1	whole body	PCBs (Aroclors), pesticides, SVOCs, metals, inorganic arsenic, butyltins, lipids, dioxins and furans (subset of samples), PCB congeners (subset of samples)	EW-Fish Collection 2008	Windward (2008d)
	2005	2	1		PCBs (Aroclors), mercury, lipids,	EW-Fish Collection 2005	Windward (2005)
Shiner surfperch	2008	8	10	whole body	PCBs (Aroclors), pesticides, SVOCs, metals, inorganic arsenic, butyltins, lipids, dioxins and furans (subset of samples), PCB congeners (subset of samples)	EW-Fish Collection 2008	Windward (2008d)
	2005	3	6 to 8		PCBs (Aroclors), mercury, lipids,	EW-Fish Collection 2005	Windward (2005)
Striped perch	1998	3	2 to 8	skinless fillet	PCBs (Aroclors), mercury, TBT, lipid	WSOU	ESG (1999)
	1998	3	2 to 8	skin-on fillet	PCBs (Aroclors), mercury, TBT, lipid	WSOU	ESG (1999)
Juvenile Chinook salmon	2009	1	165	stomach contents	metals, PAHs	EW-Chinook sampling 2009	Windward (2009 in prep)
		6	4-36	whole body	PCBs (Aroclors), pesticides, SVOCs, metals, butyltins, lipids, PCB congeners and, dioxins and furans	EW-Chinook sampling 2009	Windward (2009 in prep)
	2002	6	7 to 8		mercury	EW-Salmon	Windward (2002b)

Species	Year of Sample Collection	No. of Samples	No of Individuals per Sample	Sample Type	Analytes	Event Name	Source
Dungeness crab[b]	2008	1	7	edible meat	PCBs (Aroclors), pesticides, SVOCs, metals, inorganic arsenic, butyltins, lipids, dioxins and furans (subset of samples), PCB congeners (subset of samples)	EW-Fish Collection 2008	Windward (2008d)
				hepatopancreas	PCBs (Aroclors), pesticides, SVOCs, metals, inorganic arsenic, butyltins, lipids, dioxins and furans (subset of samples), PCB congeners (subset of samples)	EW-Fish Collection 2008	Windward (2008d)
Red rock crab[b]	2008	8	7	edible meat	PCBs (Aroclors), pesticides, SVOCs, metals, inorganic arsenic, butyltins, lipids, dioxins and furans (subset of samples), PCB congeners (subset of samples)	EW-Fish Collection 2008	Windward (2008d)
	1998	3	5		PCBs (Aroclors), mercury, TBT	WSOU	ESG (1999)
	2008	8	7	hepatopancreas	PCBs (Aroclors), pesticides, SVOCs, metals, inorganic arsenic, butyltins, lipids, dioxins and furans (subset of samples), PCB congeners (subset of samples)	EW-Fish Collection 2008	Windward (2008d)
Mussels	2008	11	89 to 101	soft tissue	PCBs (Aroclors), pesticides, SVOCs, metals, inorganic arsenic, butyltins, lipids, dioxins and furans (subset of samples), PCB congeners (subset of samples)	EW-Fish Collection 2008	Windward (2008d)
	1997	3	50 to 100		PCBs (Aroclors), SVOCs, pesticides, metals, butyltins, lipids, solids	KC WQA	King County (1999)
	1996	3	50 to 100		PCBs (Aroclors), SVOCs, pesticides, metals, butyltins, lipids, solids	KC WQA	King County (1999)
Shrimp	2008	1	26	whole body	PCBs (Aroclors), pesticides, SVOCs, metals, inorganic arsenic, butyltins, lipids, dioxins and furans (subset of samples), PCB congeners (subset of samples)	EW-Fish Collection 2008	Windward (2008d)
Clams[c]	2008	22	1 to 15	soft tissue	PCBs, TBT, mercury	EW-Clam Survey	Windward (2008c)
Benthic invertebrates	2008	13	nd	whole body	PCBs (Aroclors), PAHs, metals, butyltins, lipids	EW Benthic Survey	Windward (2008b)

Species	Year of Sample Collection	No. of Samples	No of Individuals per Sample	Sample Type	Analytes	Event Name	Source
Sand sole[d]	2005	6	1	whole body	PCBs (Aroclors), metals, lipids. solids	EW-Fish Collection 2005	Windward (2005)

[a] The results for the fillet composite samples and the remainder composite samples were weighted based on the fraction of the whole-body mass represented by each sample in order to calculate whole-body results (Windward 2006).

[b] Data from hepatopancreas composite samples will be mathematically combined with data from composite samples of edible meat to form composite samples of edible meat plus hepatopancreas. Whole-body (i.e., edible meat plus hepatopancreas) crab concentrations will be calculated using the relative weights and concentrations of the edible meat and hepatopancreas.

[c] Geoduck tissue residues will be evaluated as part of the human health risk assessment and addressed as an uncertainty in the ecological risk assessment for the benthic tissue residue line of evidence due to the lack of relevant toxicity data for the evaluation of geoduck tissue residues for the protection of geoducks. The geoducks are not a component of the diets of any of the ecological receptors.

[d] Sand sole data will be evaluated in the uncertainty assessment as a surrogate for brown rockfish data.

ERA – ecological risk assessment
ESG – Environmental Solutions Group
EW – East Waterway
KC – King County
nd – not determined
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
TBD – to be determined
TBT – tributyltin
WSOU – Waterway Sediment Operable Unit
WQA – water quality assessment

2.1.3 Surface water chemistry

Three previous sampling events in the EW have included the collection of surface water data (Table 2-4); the 1996-1997 water quality assessment (WQA) conducted by King County represents the majority of these data. As part of the WQA, King County conducted sampling on a weekly basis from October 1996 to June 1997, for a total of 192 samples. Although these data were collected more than 10 years ago, they represent a substantial dataset and, therefore, will be incorporated into the ERA following a usability analysis to determine appropriate methods for combining the existing datasets with the 2008-2009 ERA dataset (Windward 2009). The usability analysis will evaluate the similarity and differences in the variability, estimates of the mean and the upper confidence limit on the mean (UCL), and magnitude of any differences before determining how the data will be used in the ERA.

Table 2-4. Summary of available and proposed surface water data for use in the EW ERA

Year of Sample Collection	No. of Samples Analyzed	Analytes	Event Name	Source
2008-2009	49	metals (filtered and unfiltered), PCBs congeners, SVOCs, TBT, and conventionals	SRI/FS	Windward (2009)
2004-2005	36[a]	metals (unfiltered), PCBs (Aroclors), pesticides (dieldrin and DDT), TBT, and conventionals	2005 EW Water Quality Monitoring	Anchor and Windward (2005)
2000	6	metals (unfiltered), PCBs (Aroclors), pesticides (aldrin, dieldrin, DDT, and chlordane), TBT, and conventionals	2000 EW Water Quality Monitoring	SEA (2000)
1996-1997	192[a]	metals (filtered and unfiltered), SVOCs, and conventionals	King County WQA	King County (1999)

[a] Samples analyzed for conventional parameters only are not included in sample count.

ERA – ecological risk assessment
EW – East Waterway
FS – feasibility study
PCB – polychlorinated biphenyl
SRI – supplemental remedial investigation
SVOC – semivolatile organic compound
TBT – tributyltin
WQA – water quality assessment

The SRI sampling in 2008 and 2009 (Windward 2009) includes five separate sampling events, three of which have been conducted to date. The remaining two events will be conducted by March 2009. Surface water sampling for the SRI was designed to represent a variety of environmental conditions (i.e., habitats, seasons, depths, and flow rates) in the EW when combined with historical data.

2.1.4 Sediment toxicity tests

Bioassays have been conducted as part of multiple projects to characterize the toxicity of EW sediment in the biologically active zone (typically the top 10 cm) and to assess the eligibility of dredged material (typically in 1.2-m depth intervals) to be placed at open water disposal sites. Historical bioassay test results will only be included in the ERA if they were conducted with sediment collected within the top 10 cm and in accordance with Puget Sound Estuary Program (PSEP) protocols (PSEP 1995). Results from bioassays conducted for dredged material assessments will not be included in the ERA because the sediment was composited over 1.2-m depth horizon from multiple locations, and thus does not represent the benthic invertebrate exposure regime. The following bioassays have been conducted as part of historical sediment characterizations: acute (10-day) amphipod survival using the amphipods *Eohaustorius estuarius*, *Ampelisca abdita*, or *Rhepoxynius abronius;* acute (48-hour) bivalve larvae/echinoderm embryo normal survival test using the blue mussel, *Mytilus galloprovincialis*, or the sea urchin *Strongylocentrotus* spp.; and the chronic (20-day) juvenile polychaete survival and growth test using *Neanthes arenaceodentata*.

Seventy-one sediment samples from the 0- to 10-cm sediment horizon have bioassay results (Table 2-5). Most of the sediment samples were tested using three kinds of test organisms (i.e., amphipods, larval bivalves, and juvenile polychaetes). Results of toxicity tests are presented in Appendix G of the EISR (Anchor and Windward 2008).

Table 2-5. Sediment bioassays conducted in EW surface sediment

Event	Sampling Dates	Collection Method	No. of Bioassay Samples	Analytes[a]	Source
EW SRI Sampling	2009	0.1 m^2 van Veen	5	SMS; DMMP	Windward 2009
EW/Harbor Island Nature and Extent – Phases 1 and 2	2001	0.1 m^2 van Veen	41	SMS; DMMP	Windward (2002a)
T-18 – PDM	2000	0.1 m^2 van Veen	9	SMS; DMMP	Windward (2001)
King County CSO 96	1996	0.1 m^2 van Veen	6	SMS	King County (1996)
Harbor Island SRI	1995	0.1 m^2 van Veen	3	SMS	EVS (1996a, b)

[a] SMS analytes include PCBs, SVOCs, VOCs, metals (arsenic, cadmium, chromium, copper, lead, mercury, silver, and zinc), TOC, and grain size; DMMP analytes include PCBs, pesticides, SVOCs, TBT, metals (antimony, arsenic, cadmium, chromium, copper, lead, mercury, nickel, silver, and zinc), TOC, and grain size.

CSO – combined sewer overflow
DMMP – Dredged Material Management Program
PCB – polychlorinated biphenyl
SVOC – semivolatile organic compound
T-18 – Terminal 18
TBT – tributyltin

PDM – post-dredge monitoring
SMS – Washington State Sediment Management Standards
SRI – supplemental remedial investigation
TOC – total organic carbon
VOC – volatile organic compound

Additional bioassays will be conducted as part of the EW SRI. Sediment for bioassays will be collected as part of the surface sediment sampling program that will be conducted in early 2009. Three endpoints (two acute and one chronic) will be tested, according to Washington State Sediment Management Standards (SMS) requirements.

2.1.5 Sediment profile images

Visual assessment of benthic habitats and the benthic community successional stage was conducted in the fall of 2008 using a sediment profile imaging (SPI) technique whereby, a specialized camera mounted on a platform is extended into the surface sediment, and replicate photographs are taken of the sediment surface (plan view) and sediment column (profile view). These images were analyzed to characterize surface roughness, evidence of physical disturbance, apparent sediment grain size, stratification or layering within the sediment, depth of biological activity and oxygenated zone within the sediment, density of burrows or tubes and presence of wood waste or other debris.

The SPI results were evaluated to confirm the biological active zone that will be the focus of the benthic risk assessment. The depth to which the majority of the benthic organisms live is primarily evidenced by the depth of the oxygenated zone in the surface sediments. The apparent redox potential discontinuity depth (aRPD) can be used as an indication of where the majority of the biological reworking and oxygenation of the sediment by benthic organisms occurs. Within the EW the aRPD ranged from 0.2 to 5.1 cm (average = 2.1 cm). Individual benthic organisms can occur below the oxygenated zone, particularly where the physical environment is stable with minimal perturbations (e.g., periodic erosion, predation, or chemical exposure) such that mature communities can establish themselves. The benthic community present in the EW is largely a mature (Stage 3) community characterized by larger, deeper burrowing, longer-lived organisms. In almost all cases, Stage 1 organisms (earlier colonizers that are typically smaller and shorter-lived) were found at the same locations as Stage 3 organisms. These Stage 1 organisms may be indications of a late season recruitment. The extent of subsurface incursions by larger organisms was illustrated by the presence of feeding voids in the sediment column. In the EW, feeding voids were observed in approximately 20% of the images evaluated. Of those, most (76%) images had only 1 void; however, up to 5 were observed at some stations. Feeding voids ranged from 2.3 to 18.3 cm (averaged 4.1 cm) below the sediment surface; however, only 9% of the voids were > 10 cm (Windward 2009). Given the maximum depth of the aRPD (5.1 cm) and the limited incursion of organisms beyond 10 cm, the biologically active zone will be based on a depth of 10 cm as is typical of other Puget Sound sediment investigations and as suggested by Ecology guidelines for evaluating sediment benthos.

2.2 Selection of COPCs

This section describes each step in the COPC selection process for each ROC group (i.e., benthic invertebrate community, crabs, fish, and wildlife). The selection process has been adopted from the approach used for the LDW ERA and will involve an initial screening step to identify COIs, followed by the derivation and selection of TRVs from regulatory thresholds or the scientific literature for COIs. The final step will be a comparison of the maximum dose or detected chemical concentration of a COI to its TRV to identify COPCs that will be further evaluated in the ERA. The COPC selection process is outlined in Table 2-6 and described in more detail in the remainder of this section.

Table 2-6. COI and COPC selection process for the EW ERA

ROC	Exposure Pathway	COI Selection	COPC Selection
Benthic Invertebrate community	surface sediment	selected if either: ◆ there is an SQS criterion or DMMP level and the chemical is detected in any sediment sample ◆ detected in >5% of surface sediment samples (i.e., any more frequently detected chemical, regardless of availability of regulatory criterion or guideline)	COI is retained as COPC if the maximum detected concentration[a] exceeds either the SMS or toxicity-based DMMP guideline, or TRV.
	tissue	PCBs, mercury, and TBT	COI is retained as COPC if the maximum detected concentration[a] exceeds the aquatic invertebrate tissue TRV.
	porewater	selected if VOCs are detected in any porewater sample	COI is retained as COPC if the maximum detected concentration[a] exceeds the chronic AWQC/WQS.
	surface water	Selected if detected in any surface water sample	COI is retained as COPC if the maximum detected concentration in water[a] exceeds chronic WQS, AWQC, or literature TRVs.
Crabs	tissue	selected if two of the following three criteria are met: ◆ detected in >5% of surface sediment samples ◆ identified as a bioaccumulative chemical by EPA (2000a) ◆ detected in any crab tissue sample from the EW	COI is retained as COPC if the maximum detected concentration in crab tissue[a] exceeds the tissue NOAEL-based TRV for crabs or other decapods[b]
Juvenile Chinook salmon, English sole, and brown rockfish	tissue diet	selected if two of the following three criteria are met: ◆ detected in >5% of surface sediment samples ◆ identified as a bioaccumulative chemical by EPA (2000a) ◆ detected in any fish tissue or prey sample from the EW	COI is retained as COPC if the maximum detected concentration in fish tissue[a] exceeds the fish tissue NOAEL-based TRV or COI is retained as COPC if it is a PAH or a dietary metal[c] and the maximum detected concentration in diet[a, d] exceeds the dietary NOAEL-based TRV for fish.
	surface water	selected if detected in any surface water sample	COI is retained as COPC if the maximum detected concentration in water[a] exceeds chronic WQS or AWQC values, or literature TRVs.
Osprey, pigeon guillemot, river otter, and harbor seal	diet	selected if two of the following three criteria are met: ◆ detected in >5% of surface sediment samples ◆ identified as a bioaccumulative chemical by EPA (2000a) ◆ detected in any prey tissue sample from the EW	COI is retained as COPC if the maximum dietary dose[a, e] for a COI exceeds the dietary NOAEL-based TRV for birds or mammals.

[a] Detection limits exceeding the screening criteria for COPC selection will be evaluated as uncertainties in the characterization of COPC risks.

[b] TRVs based on a broader search of effects on aquatic invertebrates will be derived, if no crab- or decapod-specific TRV is available.

[c] Dietary metals include arsenic, antimony, cadmium, chromium, cobalt, copper, lead, molybdenum, nickel, silver, thallium, and zinc.

[d] The maximum concentration of a COI in the fish diet (except juvenile Chinook) will be calculated as a weighted concentration consisting of 10% of the maximum sediment concentration (to account for exposure via incidental sediment ingestion) plus 90% of the maximum prey tissue concentration. For juvenile Chinook, no incidental sediment ingestion will be assumed; therefore, their diet will be based on the maximum invertebrate concentration. Note that 10% sediment ingestion is a conservative assumption and that lower sediment ingestion rates are assumed in the exposure analysis.

[e] The maximum dietary dose of a COI for wildlife species will be calculated using the maximum concentration in prey and a site use factor of 1.

AWQC – ambient water quality criteria
COI – chemical of interest
COPC – chemical of potential concern
DMMP – Dredged Material Management Program
EW – East Waterway
NOAEL – no-observed-adverse-effect level
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
ROC – receptor of concern
SMS – Washington State Sediment Management Standards
SQS – sediment quality standard
TBT – tributyltin
TRV – toxicity reference value
VOC – volatile organic compound
WQS – water quality standard

2.2.1 COI identification

The first step of the COPC identification process is to identify COIs using a method specific to each ROC and each exposure pathway for that ROC (Table 2-6). For the surface sediment pathway for the benthic invertebrate community, any chemical detected in sediment for which a Washington State sediment standard (i.e., SQS or CSL) or Dredged Material Management Program (DMMP) guideline is available will be identified as a COI. For the remaining chemicals, those infrequently detected (< 5% detection frequency) will be mapped, and the spatial distribution and concentrations will be qualitatively evaluated for the presence of potential source areas (defined as a chemical detected in three or more adjacent samples). Chemicals with identified source areas or detection frequencies > 5% will also be identified as COIs. For the assessment of benthic invertebrate tissue, polychlorinated biphenyls (PCBs), mercury, and tributyltin (TBT) have been selected as COIs primarily because of their potential to bioaccumulate and their prevalence in sediment samples collected from the EW.[1] For the benthic invertebrate porewater pathway, any volatile organic compound (VOC) detected in porewater will be considered a COI. For the surface water pathway, any chemical detected in surface water will be considered a COI.

For the assessment of tissue for crabs and fish and the dietary exposure pathways for fish, birds, and mammals, both the detection frequency in sediment and the bioaccumulation potential of a chemical (based on both chemical properties and detection in tissue) will be considered in selecting COIs. A weight-of-evidence approach will be used, such that two of the following three criteria must be met: 1) detection in > 5% of surface sediment samples, 2) identification as a bioaccumulative chemical by EPA (2000a), and 3) detection in any tissue sample. For the surface water pathway for fish, any chemical detected in surface water will be considered a COI. This approach is consistent with the LDW ERA.

COIs will be further evaluated to select COPCs based on the magnitude of each COI relative to its TRV or screening criterion. Non-detected chemicals with detection limits exceeding the maximum detected concentration and the associated TRV or screening criterion will be evaluated in the uncertainty section.

2.2.2 TRV derivation

TRVs will be used to both select COPCs and evaluate risks to receptors. TRVs will represent thresholds (including regulatory standards) in sediment, water, or tissue that are protective of survival, growth, and reproduction of organisms in each receptor group and will address:

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[1] The LDW ERA did not select COIs for benthic invertebrate tissue, although TBT was evaluated in benthic invertebrate tissue as part of the TBT risk characterization and PCBs were evaluated in the uncertainty section.

- Predicted toxicity of sediment to benthic invertebrates from COIs (Section 3.1).
- Predicted toxicity of water to benthic invertebrates and fish from COIs.
- Tissue residues resulting in adverse effects on:
  - Benthic invertebrates (PCBs, mercury, and TBT only)
  - Crabs (all COIs) (Section 3.2)
  - Fish (all COIs except polycyclic aromatic hydrocarbons [PAHs] and dietary metals) (Section 4)
- Dietary exposures resulting in adverse effects on:
  - Fish (for PAHs, and dietary metals)
  - Wildlife (all COIs) (Section 5)

Sediment screening criteria for the protection of the benthic invertebrate community will be the SMS or DMMP guidelines, where no SMS standards exist. Other marine sediment quality guidelines or literature-based effects will be considered as TRVs if a COI that is not addressed by SMS or toxicity DMMP criteria is identified. The evaluation of surface water for effects on benthic invertebrates and fish will be based on Washington State chronic marine water quality standards (WQS; WAC 173-201A-240). If no WQS are available, national ambient water quality criteria (AWQC) will be used. Criteria will be evaluated to ensure that they are based on survival, growth, and reproduction of invertebrates and fish. Criteria based on other receptors and endpoints will not be used. Criteria based on dissolved concentrations will be used when available. If no standards or criteria are available, or if criteria are not based on relevant receptors or endpoints, then water-based effects values will be identified either from the final chronic value provided in the water quality criteria documentation or from literature sources. The same approach will be used for development of TRVs for VOCs detected in porewater.

Tissue-residue and dietary TRVs will be based on those used in the LDW ERA,[2] which were derived from relevant-effects data reported in the literature or scientific effects databases. Any literature-based TRVs selected for evaluating EW risks will be updated if any recent studies that were not published at the time the LDW TRVs were derived are identified.

If a TRV is not available from those derived for the LDW ERA because a unique COI is identified for the EW, then a literature search will be conducted for that chemical to attempt to derive a TRV following the procedure used during the LDW ERA as described below.

[2] For VOCs in porewater, TRVs from the LDW ERA will also be used and updated if more recent studies are available.

On-line databases including BIOSIS, EPA's ECOTOX database, and the US Army Corps of Engineers Environmental Residue Effects Database, which includes nearly all data from Jarvinen and Ankley (1999), will be searched for scientific literature to identify studies reporting effects to aquatic (or aquatic-dependent) organisms based on survival, growth, and reproduction.[3] Only growth and survival endpoints will be evaluated for assessing risks to juvenile Chinook salmon because egg and embryo life stages do not occur in the EW and because their exposure in the EW as adults is limited. If acceptable reproduction TRVs in which fish were exposed solely as juveniles are identified, they will be considered for juvenile Chinook salmon. Furthermore, if a TRV is selected for other fish receptors based on reproduction, the higher salmon-specific TRV will be evaluated to determine if it is protective of sublethal (i.e., growth) effects in salmon. Original sources of published toxicity data will be obtained and reviewed to verify the data summarized in the databases as well as determine the suitability of the studies for use in the ERA. The following considerations will determine the acceptability of a study for TRV derivation:

- All selected TRVs will be based on laboratory toxicological studies. Studies that exposed organisms to chemicals in the field, or that fed organisms field-collected prey for chemical exposure are not considered acceptable. Field studies will not be used to derive TRVs because adverse effects observed in organisms from field studies may be attributed to the presence of multiple chemicals and/or other uncontrolled environmental factors, rather than to a single test chemical.
- Selected TRVs will be based preferentially on dietary, sediment, or water exposure studies. Studies conducted using intraperitoneal injection, egg injection, forced ingestion, or oral gavage as exposure routes are not considered representative of the ROC exposure conditions but will be used if no other studies are available.
- Studies with egg production endpoints for chicken or quail, such as Edens and Garlich (1983) and Edens et al. (1976), are considered to be irrelevant to wild bird populations and will only be considered if data from other, more appropriate studies are not available. These data are not relevant because chickens and quail have been bred to have high egg-laying rates. Even with a significant reduction in their baseline egg production, these egg production rates may be much higher than those of any wild avian species. These differences in reproductive physiology result in high uncertainty when extrapolating a reproductive effect threshold from egg production rates for chickens or quails to avian wildlife.

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[3] The reproductive endpoint is inclusive of early life stage developmental effects (e.g., growth from egg through fry stage, embryo viability).

- Whole-body tissue TRVs for organic contaminants in fish will be estimated from egg-residue tissue TRVs using the adult-to-egg ratio for the contaminant concentration, if such data are available.
- All tissue residues must be analyzed as part of the effects studies accepted for use in TRV derivation (no nominal values will be used).
- Toxicity studies conducted with chemical forms not likely found in the EW, such as the fungicide methylmercury dicyandiamide, will not be used to develop TRVs. Toxicity of these chemical forms is not comparable to the toxicity of forms of chemicals present in the EW.

After the literature searches are conducted, all acceptable studies for TRV derivation will be compiled for each ROC and medium. The acceptable studies for each COI, ROC, and medium will be evaluated and the most appropriate TRV for the assessment endpoint associated with the ROC will be selected. The TRVs will be selected to represent both the no-observed-adverse-effect level (NOAEL) and the lowest-observed-adverse-effect level (LOAEL). The NOAEL represents the level below which adverse effects are not expected and will be used to select COPCs and assess risks. The LOAEL represents the level at which an effect has been observed; LOAEL TRVs will be used to assess risks in support of risk management decisions.

The NOAEL will be selected as the highest level below the selected LOAEL with the same endpoint. If no NOAEL with the same endpoint as the selected LOAEL is available, the NOAEL will be selected as the highest NOAEL below the selected LOAEL based on another endpoint (survival, growth, or reproduction). For COIs without NOAELs lower than the selected LOAEL, the NOAEL will be derived by applying the following uncertainty factors following EPA Region 10 guidance (EPA 1997b):

- Acute or subchronic LOAEL/10
- Chronic or critical lifestage[4] LOAEL/5
- LC50 (concentration that is lethal to 50% of an exposed population) (or similar)/50

The LOAEL will be selected from among the list of possible TRVs if it is the lowest concentration or dose at which an effect has been observed for any of the three endpoints evaluated (i.e., survival, growth, or reproduction) and a clear exposure-effect or dose-response relationship has been observed. COIs with no TRVs will be

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[4] Chronic exposure is defined as > 15% of an organism's lifespan (Calabrese and Baldwin 1993). Exposure is assumed to be chronic if the duration is greater than 10 weeks for birds and greater than 1 year for mammals (Sample et al. 1996). For fish, chronic exposure duration was assumed to be 28 days or greater. A critical life stage is one that occurs during reproduction, gestation, or development (Sample et al. 1996).

discussed in the uncertainty analysis. A final list of TRVs, identifying new or modified TRVs, will be submitted to EPA for review.

2.2.3 COPC identification

The maximum detected concentration in each exposure medium for each COI for all exposure pathways except the diet will be compared to its respective TRV; COI concentrations exceeding the TRV will be retained as COPCs. Chemicals with maximum detected concentrations below the TRV will not be evaluated further. For TRVs based on regulatory standards or guidelines, the value equivalent of a no-effect or chronic threshold will be used (i.e., sediment quality standards [SQS] or screening level [SL] for sediment; chronic water quality standards [WQS]/AWQC for water).

For the dietary exposure pathway for fish, the maximum dietary exposure concentration for each COI will be compared to the respective NOAEL TRV. The maximum dietary exposure concentrations for each ROC-COI pair will be represented by a weighted average of 10% maximum sediment concentration (to account for exposure via incidental sediment ingestion [except in the case of juvenile Chinook, which do not ingest sediment]) and 90% maximum prey tissue concentration (Equation 2-1). Although no specific data on sediment ingestion by EW fish species have been identified, 10% has been identified as an upper-bound estimate of sediment ingestion for fish with benthic diets such as English sole and shiner surfperch based on discussions with fish experts (Johnson 2006; Lange 2006).[5]

$$\text{Maximum [diet]} = \text{Maximum [sed]} \times 10\% + \text{Maximum [tissue]} \times 90\% \qquad \textbf{Equation 2-1}$$

Where:

diet = concentration in the diet (mg/kg dw)
sed = maximum sediment concentration (mg/kg dw)
tissue = maximum concentration in tissue of any prey type (mg/kg dw)

For the dietary exposure pathway for birds and mammals, the maximum dietary dose for each COI will be compared to the respective NOAEL TRV. The maximum dietary exposure dose for each ROC-COI pair will be calculated using the following equation:

$$\text{Dose} = \frac{IR_{diet} \times C_{diet}}{BW} \qquad \textbf{Equation 2-2}$$

[5] Juvenile Chinook do not consume sediment. Therefore the following equation will be used for juvenile Chinook: Maximum [diet] = Maximum [tissue]

Where:

Dose	=	COI concentration ingested per day via diet and normalized to body weight (mg/kg bw/day)
IR_{diet}	=	dietary ingestion rate (kg/day dw)
C_{diet}	=	maximum concentration in tissue of any prey type (mg/kg dw)
BW	=	wildlife species body weight (kg ww)

COIs with maximum dietary exposure concentrations or doses greater than the NOAEL TRV for ROC-COI pairs for fish, birds, or mammals will be selected as COPCs for the respective ROCs; remaining COIs will not be evaluated further.

3 Benthic Invertebrate Risk Approach

The benthic invertebrate community as a whole and crab populations were selected as ROCs to represent benthic invertebrates that may be exposed to sediment-associated chemicals in the EW. The overall approach to the benthic invertebrate risk assessment is designed to address the following risk questions:

- Are concentrations of COPCs in surface sediment at levels that might cause an adverse effect on the benthic invertebrate community in the EW?
- Are concentrations of COPCs in EW surface water at levels that might cause an adverse effect the benthic invertebrate community in the EW?
- Are concentrations of PCBs, TBT, and mercury in invertebrate tissue at levels that might cause an adverse effect on the benthic invertebrate community in the EW?
- Are concentrations of COPCs in crab tissues at levels that might cause an adverse effect on the crab community in the EW?

Lines of evidence that address the risk questions for assessing benthic invertebrates are presented in Table 2-1.

3.1 BENTHIC INVERTEBRATE COMMUNITY

The risk assessment for benthic invertebrates will be based on measures of exposure to COPCs and measures or predictions of effects on the benthic community using various lines of evidence. Risks will be initially characterized based on each line of evidence and then combined using a weight-of-evidence approach to characterize the occurrence and magnitude of risks associated with sediment COPCs. This section describes each of these components.

3.1.1 Exposure assessment

Exposure of the benthic invertebrate community to COPCs in the EW will be based on an evaluation of COPC concentrations in surface sediment, surface water, seep water, and selected COPCs in benthic invertebrate tissue.

Surface sediment data for COPCs will be used to characterize the exposure regime for the benthic invertebrate community. Surface sediment will be defined as the top 10 cm of the sediment column, which is representative of the typical biologically active zone used by benthic invertebrates in Puget Sound (Ecology 2008) and was confirmed by site-specific SPI data.

Dry-weight sediment concentrations for non-ionic organic compounds will be normalized to TOC according to SMS rules if the sediment TOC falls between 0.5 and 4.0% (Michelsen and Bragdon-Cook 1993; Bragdon-Cook 1995). Higher or lower TOC will result in use of the apparent effects thresholds (AETs) that formed the basis of the SMS and are expressed on a dry-weight basis.

Benthic invertebrate exposure to COPCs in sediment will be assessed on an individual sample basis, because most benthic invertebrates are relatively sedentary. An evaluation of the spatial scale of the distribution of COPCs in sediment will be provided as part of the risk characterization to address community-level effects and will be discussed in the uncertainty section of the ERA because of the role of benthic invertebrates as prey for other wider-ranging ROCs.

Benthic invertebrates were collected for tissue analysis in October 2008;[6] specifically, composite samples of small benthic organisms living in specific areas of the waterway represent available tissue. These tissue composites will provide a chemical-specific assessment of exposure of benthic invertebrates to PCBs, TBT, and mercury.

Surface water data for COPCs will be used to further characterize the exposure regime for the benthic invertebrate community in EW. The EW is tidally influenced, which tends to integrate water exposures over time. Benthic invertebrate exposures to water-borne contaminants will be assessed on both a site-wide and localized area basis. EPCs will be calculated from a combined water dataset (new and existing), following an evaluation of the comparability of the historical water quality data. The site-wide exposure point concentration (EPC)

[6] Fall sampling represents a reasonable sampling period for invertebrate tissue residues because of the peak in abundance and biomass that occurs following summer recruitment and growth. Historical data from Nichols (1975) and Word et al. (1983) showed maximum annual infaunal total abundance occurring in October. Alden et al. (1997) reports maximum infaunal biomass in fall and winter in the Chesapeake Bay estuary.

will be calculated as the 95% UCL on the mean (or maximum concentration, if the number of detected concentrations is <6) of all data using ProUCL 4.0.[7]

Porewater exposures will be evaluated on a sample-specific basis. Benthic invertebrates in localized areas will be assumed to have been exposed to volatile organic compounds (VOCs), if these chemicals are detected in porewater collected from intertidal seep samples at concentrations above their respective WQS or AWQC.

3.1.2 Effects assessment

Effects on the benthic community will be estimated based on comparison to TRVs. Promulgated sediment standards, regional sediment guidelines, marine water quality standards or national criteria (in the case of VOCs in porewater) and effects-based tissue residue TRVs will be used in the assessments for the benthic community. The process for deriving TRVs is presented in Section 2.2.2.

The SMS will be used to assess effects on the benthic community; toxicity-based DMMP regional guidelines for sediment quality will be used where no SMS are available. The SMS are numerical chemical standards based on AETs developed for the PSEP (Barrick et al. 1988). An AET is the highest "no effect" chemical-specific sediment concentration above which a significant adverse biological effect always occurred among the several hundred samples used in its derivation. AETs were empirically derived using data from field-collected sediment samples that contained diverse chemical mixtures analyzed simultaneously for chemistry and toxicity. The data used to derive the 1988 AETs were collected from various locations in Puget Sound between March 1982 and September 1986. AETs for four endpoints (i.e., amphipod mortality, abnormal development of oyster larvae, depressions in benthic invertebrate community abundance, and changes in bacterial bioluminescence [Microtox®] relative to reference samples) were developed for 47 chemicals. In general, the lowest AET for each chemical has been identified as the SQS; the second lowest AET has been identified as the cleanup screening level (CSL). A concentration below the SQS corresponds to a sediment quality that will result in no adverse effects to biological resources; concentrations between the SQS and the CSL correspond to a sediment quality that will result in minor adverse effects (Washington Administrative Code [WAC] 173-204); exceedance of the CSL equates to more severe effects.

The AETs were also used as the basis for the development of decision thresholds for the management of dredged material. The SL is the approximate equivalent of the SQS (sediment with chemistry below the SL is considered suitable for open water disposal); whereas, sediment concentrations above the maximum level (ML)

[7] ProUCL 4.0 assesses the distributional characteristics of a dataset, including non-detects (which are identified as such and represented by the reporting limit), and then selects the best method for calculating a UCL based on those attributes (EPA 2007b).

are considered unsuitable. The suitability of material with concentrations falling between the SL and ML are further evaluated through direct toxicity testing.

Promulgated standards and regional guidelines for sediment are summarized in Table 3-1. If no standards or regional guidelines are available, TRVs will be developed from the literature (see Section 2.2.2 for a description).

Table 3-1. Biological effect endpoints used to determine the SMS and toxicity-based DMMP guidelines

Chemical	Unit	SQS	CSL	SL	ML	Biological Endpoint Used to Establish SQS/SL	Biological Endpoint Used to Establish CSL/ML
Metals							
Antimony	mg/kg	na	na	150	150	community abundance	community abundance
Arsenic	mg/kg	57	93			community abundance	amphipod mortality
Cadmium	mg/kg	5.1	6.7			community abundance	amphipod mortality
Chromium	mg/kg	260	270			community abundance	amphipod mortality
Copper	mg/kg	390	390			oyster abnormality; Microtox®	oyster abnormality; Microtox®
Lead	mg/kg	450	530			community abundance	Microtox®
Mercury	mg/kg	0.41	0.59			Microtox®	oyster abnormality
Nickel	mg/kg	na	na	140	140	amphipod mortality; community abundance	amphipod mortality; community abundance
Silver	mg/kg	6.1	6.1			amphipod mortality	amphipod mortality
Zinc	mg/kg	410	960			community abundance	amphipod mortality
PAHs							
2-Methylnaphthalene	mg/kg OC	38	64			oyster abnormality; Microtox	community abundance
Acenaphthene	mg/kg OC	16	57			oyster abnormality	community abundance
Acenaphthylene	mg/kg OC	66	66			amphipod mortality; community abundance	amphipod mortality; community abundance
Anthracene	mg/kg OC	220	1,200			community abundance	amphipod mortality
Benzo(a)anthracene	mg/kg OC	110	270			oyster abnormality	amphipod mortality
Total benzofluoranthenes	mg/kg OC	230	450			oyster abnormality	amphipod mortality
Benzo(a)pyrene	mg/kg OC	99	210			oyster abnormality	amphipod mortality
Benzo(g,h,i)perylene	mg/kg OC	31	78			oyster abnormality	amphipod mortality
Chrysene	mg/kg OC	110	460			oyster abnormality	amphipod mortality
Dibenzo (a,h)anthracene	mg/kg OC	12	33			na	Microtox®
Fluoranthene	mg/kg OC	160	1,200			oyster abnormality	community abundance
Fluorene	mg/kg OC	23	79			oyster abnormality	community abundance
Indeno (1,2,3,-c,d)pyrene		34[a]	88			oyster abnormality	amphipod mortality
Naphthalene	mg/kg OC	99	170			oyster abnormality	community abundance

Chemical	Unit	SQS	CSL	SL	ML	Biological Endpoint Used to Establish SQS/SL	Biological Endpoint Used to Establish CSL/ML
Phenanthrene	mg/kg OC	100[b]	480			oyster abnormality	community abundance
Pyrene	mg/kg OC	1,000	1,400			amphipod mortality	community abundance
HPAH	mg/kg OC	960	5,300			oyster abnormality	amphipod mortality
LPAH	mg/kg OC	370	780			oyster abnormality	community abundance
Phthalates							
Bis(2-ethylhexyl) phthalate	mg/kg OC	47	78			Microtox®	amphipod mortality
Butyl benzyl phthalate	mg/kg OC	4.9	64			Microtox®	community abundance
Diethyl phthalate	mg/kg OC	61	110			community abundance	amphipod mortality
Dimethyl phthalate	mg/kg OC	53	53			amphipod mortality	community abundance
Di-n-butyl phthalate	mg/kg OC	220	1,700			Microtox®	community abundance
Di-n-octyl phthalate	mg/kg OC	58	4,500			amphipod mortality	community abundance
Other SVOCs							
1,2-Dichlorobenzene	mg/kg OC	2.3	2.3			oyster abnormality	community abundance and Microtox®
1,2,4-Trichlorobenzene	mg/kg OC	0.81	1.8			Microtox®	amphipod mortality
1,4-Dichlorobenzene	mg/kg OC	3.1	9			oyster abnormality	amphipod mortality
2-Methyl phenol	µg/kg dw	63	63			amphipod mortality; oyster abnormality	amphipod mortality; oyster abnormality
2,4-Dimethylphenol	µg/kg dw	29	29			oyster abnormality	Microtox®
4-Methylphenol	µg/kg dw	670	670			oyster abnormality	Microtox®
Benzoic acid	µg/kg dw	650	650			oyster abnormality	community abundance, Microtox®
Benzyl alcohol	µg/kg dw	57	73			Microtox®	oyster abnormality
Dibenzofuran	mg/kg OC	15	58			oyster abnormality	community abundance
Hexachlorobenzene	mg/kg OC	0.38	2.3			community abundance	Microtox®
Hexachlorobutadiene	mg/kg OC	3.9	6.2			Microtox®	amphipod mortality
n-Nitrosodiphenylamine	mg/kg OC	11	11			community abundance	community abundance
Pentachlorophenol	µg/kg dw	360	690			amphipod mortality	community abundance
Phenol	µg/kg dw	420	1,200			oyster abnormality	amphipod mortality, community abundance, and Microtox®
Pesticides							

Chemical	Unit	SQS	CSL	SL	ML	Biological Endpoint Used to Establish SQS/SL	Biological Endpoint Used to Establish CSL/ML
Total DDTs (sum of 4,4'-DDE, 4,4'-DDD, and 4,4'-DDT)	µg/kg dw			6.9	69	na[c]	community abundance
PCBs							
Total PCBs	mg/kg OC	12	65			Microtox®	community abundance

Note: DMMP guideline is only listed when no SMS value is available.

Source: Washington State Sediment Management Standards (WAC 173-204); Barrick et al. (1988).

[a] The SQS for indeno(1,2,3,-c,d)pyrene is 34 mg/kg OC; the lowest AET, based on oyster abnormality, is 33 mg/kg OC.

[b] The SQS for phenanthrene is 100 mg/kg OC; the lowest AET, based on oyster abnormality, is 120 mg/kg OC.

[c] The SL is 10% of the ML.

AET – apparent effects threshold
CSL – cleanup screening level
dw – dry weight
HPAH – high-molecular-weight polycyclic aromatic hydrocarbon
LPAH – low-molecular-weight polycyclic aromatic hydrocarbon
ML – maximum level
na – not available
OC – organic carbon
PAH – polycyclic aromatic hydrocarbon
PCB – polychlorinated biphenyl
SL – screening level
SQS – sediment quality standards
SVOC – semivolatile organic compound
WAC – Washington Administrative Code

As another measure of effects, site-specific toxicity tests will be conducted on a subset of surface sediments collected from the EW. The methods for determining significant ($p < 0.05$) toxicity responses are summarized in Table 3-2.

Table 3-2. SMS biological effects criteria for marine sediment toxicity tests

Toxicity Test	Biological Effects Criteria	
	SQS	CSL[a]
Amphipod	mean mortality > 25% on an absolute basis, and significantly higher than the reference area mortality ($p \le 0.05$)	mean mortality greater than the reference area mortality plus 30%, and significantly higher ($p \le 0.05$)
Juvenile polychaete	mean individual growth rate < 70% of that of the reference area growth rate and significantly lower ($p \le 0.05$)	mean individual growth rate < 50% of the reference area growth rate and significantly lower ($p \le 0.05$)
Bivalve larvae	mean normal survival < 85% of the reference area normal survival and significantly lower ($p \le 0.10$)	mean normal survival < 70% of the reference area survival and significantly lower ($p \le 0.10$)

[a] An exceedance of the SQS biological effects criteria in any two toxicity tests at one location is considered a CSL exceedance at that location (WAC 173-204-420(3)).

CSL – cleanup screening level

SMS – Washington State Sediment Management Standards

SQS – sediment quality standards

WAC – Washington Administrative Code

3.1.3 Risk characterization

The occurrence and magnitude of effects associated with exposure of benthic invertebrates to chemical concentrations in sediment will be based primarily on predicted toxicity or the results of sediment bioassays that assess survival and growth of three benthic invertebrate species. Predicted toxicity will be based on a comparison to the SQS and CSL (or related regulatory guidelines, as appropriate), which are toxicity-based sediment standards derived from a similar suite of toxicity tests and endpoints.

Tissue residues of PCBs, TBT, and mercury in benthic invertebrates that exceed their respective tissue TRVs will be used as a secondary line of evidence for potential effects on benthic invertebrates from these two chemicals. For TBT, the tissue-residue approach will be the primary line of evidence because no sediment standard or guideline is available.

Potential effects associated with exposure to chemical concentrations in water will be based on exceedances of WQS, AWQC, or water-based TRVs that address the survival, growth, and reproduction of aquatic invertebrates. Both individual samples and a site-wide EPC for individual COPCs will be evaluated. Individual seep sample exceedances will be used to indicate potential localized effects on receptors. In the risk conclusion section of the benthic risk characterization, results will be presented in a table that summarizes risk for each ROC/COPC pair.

Uncertainties in risk conclusions for benthic invertebrates will be presented in the risk characterization section, as well as in a separate uncertainty section that summarizes overall uncertainties in the entire risk assessment.

3.1.3.1 Toxicity testing

Risks to the benthic invertebrate community were assessed by performing three bioassays on surface sediment collected in the EW. Testing was tiered, such that bioassays were conducted following sediment chemical analysis. Selected sediment samples with at least one detected chemical concentration exceeding the SQS were considered for toxicity testing,[8] which included:

- Acute 10-day amphipod mortality test
- Acute 48-hour bivalve larval or echinoderm embryo normal survival test
- Chronic 20-day juvenile polychaete survival and growth test

If no toxicity tests are conducted for a specific sediment sample, the chemical results will be used to represent the likelihood of toxicity at that location. The specific test species for the embryo/larval test will be selected based on the timing of the bioassays (e.g., oyster larvae are typically not available in winter, but mussels can be induced to spawn during that season). The results from the three sediment bioassays will be evaluated using the SMS rules for marine bioassays (Ecology 2008) (see Table 3-2), which requires comparison to toxicity results for reference sediment. Reference sediment is selected based on general grain size classes from areas of known sediment quality in Puget Sound. Typically, Carr Inlet sediment is used to represent reference conditions in Puget Sound and will be used for the EW bioassays. Because SMS criteria are based on testing of sediment samples with a mixture of chemicals from a large number of Puget Sound locations, site-specific bioassays will either confirm or overrule the SMS designation based on sediment chemistry.

3.1.3.2 Predicted risks

Predictions of risks to the benthic community will be based on comparisons of sediment COPC concentrations to the SMS, toxicity-based DMMP regional guidelines, or TRVs derived for the project (Table 3-1). COPCs in surface sediment will be compared to both the SQS and CSL. Chemicals with concentrations exceeding SMS criteria, toxicity-based DMMP guidelines, or TRVs will be identified as sediment COCs which may pose a potential risk to the benthic invertebrate community. The magnitude of risk will be discussed in terms of exceedances factors relative to SQS and CSL criteria and the areal extent of those exceedances.

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[8] Testing was decided with EPA in consultation with the natural resource trustees. In general, sediment with chemical concentrations above the SQS but below the CSL were candidates for toxicity testing. Other considerations for selecting samples for toxicity testing included proximity to a potential source and the particular chemicals exceeding SQS or CSL.

COPCs for which no TRVs are available will be discussed in the uncertainty analysis. COPCs that were never detected but have reporting limits that exceed the SMS criteria or DMMP regional guidelines will also be discussed in the uncertainty analysis.

3.1.3.3 Tissue residue

The evaluation of effects associated with COPCs in tissue will rely on comparisons of COPC concentrations in tissue to tissue-based NOAEL and LOAEL TRVs. HQs will be calculated as the tissue concentration divided by the TRV. HQs greater than 1.0 based on the LOAEL TRV will be used as evidence of potential risks to benthic receptors and to identify COCs for tissue.

3.1.3.4 Surface water concentrations

The evaluation of effects associated with COPCs in water will rely on comparisons of COPC concentrations in water representing the entire site (i.e., 95% UCL using all data in the combined dataset) to water quality criteria and effects-based TRVs. An exceedance of criteria values or TRVs will be used as evidence of potential risks to benthic receptors and to identify COCs for water. COPC concentrations in water samples at each water sampling location will also be compared to TRVs. The results of this evaluation will be provided to EPA prior to the submittal of the draft ERA and the results will be used to finalize the approach for identifying COCs for water for benthic invertebrates.

3.1.3.5 Weight-of-evidence approach

Risk conclusions will be made for each line of evidence for the assessment of risk to the benthic community and the lines of evidence will be used collectively, to draw overall conclusions that can inform risk management decisions for the site. Chemicals contributing to risk will be identified based on the:

- Magnitude of HQ exceedance, number and extent of criteria exceedances, and toxicity test results
- Spatial and temporal distribution
- Degree of uncertainty regarding exposure and effects

A weight-of-evidence approach will be applied if there is disagreement among multiple lines of evidence. In the case of the benthic invertebrates there may be disagreement between the assessments of exposure to sediment and surface water, and the assessment based on tissue residues. In these instances the uncertainties associated with both the assessment of exposure and the assessment of effects will be examined in order to identify the lines of evidence with the greatest degree of certainty. Both the strength of the exposure data and the uncertainties associated with effects data would be evaluated.

3.2 Crab Populations

Although cancrid crabs are members of the benthic community, their populations will be evaluated as a separate ROC because of their larger size, greater mobility, larger home range, and the fact that they have higher trophic status than most other invertebrates living in the sediment.

3.2.1 Exposure assessment

Cancrid crabs have been collected from throughout the EW, and tissue composites representative of the entire waterway are being analyzed for the full suite of chemicals. Each tissue composite includes six to seven individual crabs of the same species randomly selected from the total catch in the waterway. Because crab data will also be used in the HHRA, composited edible meat and the hepatopancreas are being analyzed separately. For the ERA, a total body burden will be created by summing the muscle and hepatopancreas chemical results for a given composite, based on relative weights of these individual tissues (i.e., a weighted sum will be created). The hepatopancreas represents the largest organ in a whole-body crab and is relatively high in lipids; thus, the hepatopancreas is considered a conservative estimate of organ tissue burdens. Total body burdens will be averaged and the 95% upper confidence limit on the mean (UCL) of the site-wide dataset will be used to assess crabs exposed to EW COPCs. When the detection frequency is > 5% or if there are six or more detected concentrations for a given COPC, a UCL will be calculated using ProUCL 4.0. One 95% UCL, or EPC, will be calculated for each COPC for crabs. As recommended by EPA, a 95% UCL will not be calculated if the detection frequency is less than or equal to 5% or if there are fewer than six detected values (EPA 2007b). Instead, the maximum concentration will be used as the EPC, and the uncertainty in this value will be discussed.

3.2.2 Effects assessment

Effects on crabs will be established based on a comparison to tissue-residue TRVs for crab COPCs. TRVs developed for the LDW will be used as the basis of this assessment. If applicable, TRVs will be updated using recent literature. A literature search will be conducted to develop crab tissue-residue TRVs, as described in Section 2.3.2. As with the LDW TRVs, the search will focus on chemical tissue-residue data associated with effects on decapods; however, effects data for other marine invertebrates may be considered if no crab or decapod-specific data are available. For each COPC, a TRV will be selected for both the NOAEL and the LOAEL. Chemicals with no TRVs will be discussed in the uncertainty analysis.

3.2.3 Risk characterization

Potential risks to crab populations from exposures to COPCs will be assessed by comparing the site-wide EPC for each COPC (calculated as the 95% UCL, where possible) to the decapod-specific NOAEL and LOAEL TRVs (where available).

Tissue concentrations that exceed the LOAEL will be used to identify COCs for crabs. In the risk conclusion section of the crab population risk characterization, results will be presented in a table that summarizes risk for each COPC. Uncertainties in risk conclusions for crab populations will be presented in the risk characterization section, as well as in a separate uncertainty section that summarizes uncertainties for the entire risk assessment.

The uncertainties in the tissue-residue risk characterization for crabs will be discussed as part of the risk characterization.

4 Fish Risk Approach

The overall approach to the fish risk assessment is designed to address the following risk questions:

- Are concentrations of PAH and dietary metal (arsenic, antimony, cadmium, chromium, cobalt, copper, lead, molybdenum, nickel, silver, thallium, zinc) COPCs in the diet of fish that forage in the EW at concentrations that might cause an adverse effect on benthivorous, planktivorous, or upper trophic level fish populations, or individual juvenile anadromous salmon in the EW?
- Are concentrations of COPCs in EW surface water at concentrations that might cause an adverse effect on benthivorous, planktivorous, or upper trophic level fish populations, or individual juvenile anadromous salmon in the EW?
- Are concentrations of organic chemical (except PAHs) and TBT, mercury, and selenium COPCs in the tissues of fish that forage in the EW at concentrations that might cause an adverse effect on benthivorous, planktivorous, or upper trophic level fish populations, or individual juvenile anadromous salmon in the EW?

As indicated in the above risk questions, risk to fish will be evaluated using three approaches: tissue residue, dietary exposure, and exposure to water. The critical tissue-residue approach will be used to evaluate risks from exposure to all organic COPCs except PAHs, and mercury, selenium, and butyltins. An exposure media (i.e., diet and water) approach rather than a tissue-residue approach will be used for dietary metals (arsenic, antimony, cadmium, chromium, cobalt, copper, lead, molybdenum, nickel, silver, thallium, zinc) and PAHs because they are highly regulated or metabolized by fish and, therefore, whole-body tissue residues do not accurately approximate chemical concentrations at the site of toxic action. For these reasons, the EPA metals risk assessment framework cautions against using a tissue-residue approach for metals other than organo-selenium and methylmercury (EPA 2007a). A tissue-residue approach will also be applied to butyltins because research indicates that residue-effects relationships are reliable for this class of organometals (Meador 2006). Most aquatic organisms have specific mechanisms for uptake, internal transport, sequestration, and depuration of metals (Meyer et al. 2005).

Essential metals are regulated because they are necessary for normal metabolic function, whereas other metals appear to be regulated because they mimic essential elements and are transported by the same mechanisms (Bury et al. 2003, as cited in Meyer et al. 2005). This section describes three components for the fish risk assessment: the exposure assessment, the effects assessment, and risk characterization.

4.1 EXPOSURE ASSESSMENT

Exposure of fish to COPCs in the EW will be addressed using three approaches: tissue residue, dietary, and water exposure. Direct sediment contact pathways were also identified in the conceptual site model and were determined to be of unknown significance for English sole and brown rockfish and insignificant for juvenile Chinook salmon. The direct sediment contact pathway will not be quantitatively analyzed but will be discussed in the uncertainty section of the ERA for English sole and brown rockfish for chemicals not analyzed using the tissue-residue line of evidence.

4.1.1 Tissue-residue approach

A tissue residue approach will be used for mercury, TBT, and selenium and all organic chemicals (except PAHs) because tissue concentrations of these chemicals reflect exposures from all pathways (e.g., direct sediment contact, water contact, and diet), and tissue concentrations are more reflective of the concentration at the site of action than concentrations in exposure media.

For each ROC-COPC pair established in the COPC selection process (Section 2.3.3), one site-wide EPC will be calculated as the UCL of all samples. ProUCL will be used to calculate each UCL as described in Section 3.2.1. Site-wide EPCs are appropriate measurement endpoints consistent with the English sole and brown rockfish population-level assessment endpoints identified in the problem formulation (Section 2). Site-wide EPCs are appropriate for juvenile Chinook salmon because Chinook are exposed throughout the EW during outmigration. Because individual brown rockfish foraging areas are smaller than the EW, there is uncertainty as to whether site-wide risks are reflective of risks to a subset of the EW rockfish population. To assess exposure on a smaller scale, COPC concentrations in individual rockfish samples will also be evaluated to determine if exposures may be elevated above TRVs for a subpopulation of EW rockfish at a smaller than site-wide scale.

For dioxins, furans, and dioxin-like PCBs, risks to fish will be evaluated using a toxic equivalent (TEQ) approach. Using this approach, the potencies of specific individual dioxin and furan congeners and dioxin-like PCB congeners relative to that of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD) are quantified using toxic equivalency factors (TEFs). The concentrations of each specific congener are multiplied by the congener-specific TEF, and the products are then summed to calculate a TEQ for multiple congeners in each sample, as follows:

$$TEQ = \sum_{i=1}^{n} C_i TEF_i$$

Equation 4-1

Where:

TEF_i = the TEF for an individual dioxin, furan, or dioxin-like PCB congener
C_i = Tissue concentration of an individual congener

Using the assumption that the combined effect of these dioxin, furan, and PCB congeners is additive, the total TCDD-like toxicity of the congeners is estimated by summing the TEF-concentration products for individual congeners.

The TEFs that will be used in the EW ERA for fish are those developed by the World Health Organization (WHO) 1998 (Van den Berg et al. 1998; 2006). When the dioxin, furan, or PCB congener concentrations are reported as undetected, then the TEF will be multiplied by half the reporting limit.

4.1.2 Dietary approach

The dietary approach will be used to estimate exposure to fish through the diet for PAH and exposure-media metal COPCs. The primary exposure routes of COPCs in the EW to fish are assumed to be ingestion of food and ventilation of water. Incidental sediment ingestion also has been considered as a complete exposure pathway only for English sole and brown rockfish and will be addressed as part of the dietary exposure analysis.

For chemicals evaluated using a exposure-media approach, exposure concentrations of COIs in fish diets (expressed as mg/kg dw) will be compared with dietary exposure concentrations reported in the literature (expressed as mg/kg dw). An alternative way of expressing exposure that explicitly considers an ROC's rate of chemical uptake involves the calculation of "dietary dose" expressed as µg/g fish/day. Because fish prey consumption is variable, use of a dietary dose approach is becoming more prevalent as a way to normalize dietary exposure among species (e.g., Clearwater et al. 2002). This method could also be used to predict a total dose from both water and dietary exposure, although little progress has been made in this regard (e.g., Borgmann et al. 2005). Because use of a dose-based approach for the purpose of estimating effects from dietary exposure is in its infancy, components of dose (such as ration size, feeding frequency, and food wastage) are often not reported in toxicity papers. Therefore, it is difficult to estimate accurate doses from available effects data. In addition, daily food consumption rates are not standardized for fish species as they are for wildlife, making fish dietary dose exposure calculations uncertain. Therefore, consistent with the approach used in the LDW-ERA, a dietary concentration approach rather than a dietary dose approach will be used as the dietary approach for fish exposure and effects calculations in this ERA.

The dietary exposure approach requires an approximation of the COPC concentration in an ROC's diet. To approximate the dietary concentrations for each fish ROC, the feeding habits of each ROC were considered. This section presents the methods for calculating the dietary COPC concentrations and the dietary exposure assumptions for each fish ROC.

4.1.2.1 Calculation of dietary COPC concentrations

COPC concentrations in the diet of each fish ROC will be calculated as the weighted average of COPC concentrations in sediment and prey tissue using Equation 4-2.

$$C_{diet} = \sum_{i=1}^{n} X_i C_i \qquad \textbf{Equation 4-2}$$

Where:

C_{diet} = COPC concentration in the diet (mg/kg dw)
X_i = proportion of a particular prey item (or sediment) in the diet (unitless)
C_i = COPC EPC in the prey item (or sediment) (mg/kg dw)
n = number of dietary items

The relative proportions of each prey item (or sediment) in each fish ROC's diet are summarized in Table 4-1. Rationale for these prey fractions and further ROC-specific exposure assumptions are described in the following sections for each ROC.

Table 4-1. Proportions of dietary items in dietary exposure estimates for each fish ROC

ROC	Prey Item	Proportion in Diet (unitless)	Exposure area	Source of Dietary Information
Juvenile Chinook salmon	benthic invertebrates	1	nearshore areas throughout the EW	Windward (2004), Cordell et al. (1997)
English sole	benthic invertebrates	0.99	EW-wide	Fresh et al. (1979), Wingert et al. (1979)
	sediment	0.01		Johnson (2006), Lange (2006)
Brown rockfish	benthic invertebrates[a]	0.09	EW-wide and individual mussel sampling locations	Wingert et al. (1979)
	coonstripe shrimp[b]	0.48		
	crabs	0.04		
	shiner surfperch	0.39		
	sediment[a]	0.01		Lange (2006)

[a] Individual benthic invertebrate, mussel, and sediment sample data will be used for small-scale EPC calculation to determine if there are smaller areas within the EW where rockfish exposure is elevated above TRVs.

[b] The shrimp sample was collected from throughout the EW, however the mass was limited and the sample was analyzed for PCBs, metals and SVOCs only. COPC data from individual mussel samples will be used as a surrogate for shrimp in small-scale EPC calculations.

COPC – chemical of potential concern

EPC – exposure point concentration

EW – East Waterway

ROC – receptor of concern

TRV – toxicity reference value

4.1.2.2 Juvenile Chinook salmon

Stomach contents analyses of juvenile Chinook salmon from the LDW indicate that juvenile Chinook salmon typically ingest benthic invertebrates such as amphipods, worms, and clam siphons,[9] as well as drift organisms and zooplankton (Cordell et al. 1997, 1999, 2001). For the purpose of the ERA, juvenile Chinook salmon will be assumed to ingest only benthic invertebrates. Because benthic invertebrates live in close contact with sediment, they have a greater potential for sediment exposure than do other juvenile Chinook salmon prey items; therefore, this exposure assumption is conservative (i.e., may overestimate exposures but unlikely to underestimate them). Juvenile Chinook from the LDW were found to have no appreciable amounts of sediment in their stomachs (Cordell 2001); therefore, they are assumed to have no incidental sediment ingestion. Juvenile Chinook salmon generally do not use deep-water habitats (Tabor et al. 2004); therefore, exposure will be estimated assuming juvenile Chinook salmon are exposed primarily in nearshore areas. Benthic invertebrates were collected throughout the EW. The entire EW benthic invertebrate dataset will be used to calculate one EPC for each COPC for the dietary approach for juvenile Chinook salmon. The EPCs will be calculated as the UCL of the EW benthic invertebrate dataset using ProUCL as discussed in Section in Section 3.2.1. The dietary concentration for each COPC will be calculated using the site-wide benthic invertebrate EPC for that COPC in Equation 4-2. In addition to benthic invertebrate tissue data, exposure may also be separately estimated based on the chemical analysis of stomach contents of juvenile Chinook salmon collected from the EW.

4.1.2.3 English sole

Stomach contents analyses of English sole collected from Puget Sound show that English sole almost exclusively ingest benthic invertebrates such as polychaetes, amphipods, bivalves, and other mollusks (Fresh et al. 1979; Wingert et al. 1979). Based on these stomach contents analyses, all prey of English sole are assumed to be represented by the benthic invertebrate tissue chemistry data. The same benthic invertebrate site-wide EPCs developed for juvenile Chinook salmon will be used for English sole (i.e., the UCL of entire EW benthic invertebrate dataset). In addition,

[9] EW clam data will not be included in exposure calculations because benthic invertebrate samples include clams less than 2.0 cm, which are assumed to represent clam tissues consumed by fish ROCs.

incidental sediment ingestion of 1% is assumed based on anecdotal stomach contents observations of English sole and other bottom-feeding fish (Johnson 2006; Lange 2006). Site-wide EPCs will also be calculated for each COPC in sediment as the UCL of the entire EW sediment dataset. Site-wide dietary EPCs are appropriate measurement endpoints consistent with the English sole population-level assessment endpoint identified in the problem formulation (Section 2). For each COPC, one EPC will be calculated in benthic invertebrate tissue and one in sediment. These EPCs will be used in Equation 4-2 to calculate the dietary COPC concentrations.

4.1.2.4 Brown rockfish

Stomach contents analyses of brown rockfish collected from Puget Sound show that they ingest primarily shrimp and small fish, and smaller amounts of crabs and benthic invertebrates such as amphipods and isopods (Wingert et al. 1979). Respective percentages of these items in brown rockfish diet are 48, 39, 8, and 4% based on an analysis of the index of relative importance (IRI) (Wingert et al. 1979).[10] Incidental sediment ingestion of 1% of the total diet has been assumed based on the primarily epifaunal diet of the brown rockfish (Lange 2006). Shiner surfperch have been selected as representative prey fish because they are numerically dominant in the fish community in the EW, and surfperch are important prey for brown rockfish (Matthews 1990). Shiner surfperch also have a primarily benthic diet so they represent bioaccumulation of sediment-associated chemicals through the food chain (Wingert et al. 1979; Fresh et al. 1979; Miller et al. 1977). Available crab data are from crabs larger than those consumed by rockfish, but they are expected to have metal and PAH COPC concentrations similar to or higher than those of smaller crabs.

One site-wide EPC will be calculated for each COPC in each prey item for brown rockfish (benthic invertebrates, shrimp, crab, and perch) and one site-wide EPC will be calculated for each COPC for sediment. The EPC for each prey item or sediment will be estimated as the UCL over all prey or sediment samples from throughout the EW. ProUCL will be used to calculate the UCL as described in Section 3.2.1. One exception will be shrimp, which will be represented by a single sample. The dietary COPC concentrations will be calculated using the site-wide EPCs in Equation 4-2. Site-wide dietary EPCs are appropriate measurement endpoints consistent with the brown rockfish population level assessment endpoint identified in the problem formulation (Section 2). Benthic invertebrate and sediment UCLs will be combined to calculate EPCs as discussed in Section 4.1.2.4.

Because individual brown rockfish foraging ranges are smaller than the EW, and the foraging ranges of rockfish prey may also be smaller than the EW, smaller scale exposures would not be evaluated using a site-wide approach. Therefore, dietary

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[10] IRI is a metric used to determine dietary importance of food items, including numerical abundance, biomass, and frequency of occurrence in diet.

COPC concentrations will be calculated for smaller areas of the EW in addition to COPC concentrations calculated on a site-wide basis. Rockfish dietary items representative of exposure in small areas of the EW include coonstripe shrimp, benthic invertebrates, and sediment. No information on coonstripe shrimp home ranges was identified; however, based on anecdotal observations, they are likely to be similar to those of brown rockfish (Jensen 2008). Infaunal benthic invertebrate home-ranges are also likely to be smaller than those of brown rockfish, whereas shiner surfperch and crab foraging ranges are expected to be similar in size to the entire EW (Pauley et al. 1988). Because only a single site-wide shrimp composite has been collected, small-scale dietary COPC concentrations will be calculated by substituting individual mussel sample data as a surrogate for the shrimp sample data and a dietary COPC concentration will be calculated for each of the 11 mussel sample locations. At each location, the EPC concentration for each COPC in each prey item and in sediment will be calculated as follows:

- **Benthic invertebrates –** the concentration in the individual composite benthic invertebrate sample collected closest to the mussel location
- **Mussels –** the concentration in the individual composite mussel sample at that location
- **Shiner surfperch –** the site-wide EPC calculated as the UCL of the entire EW shiner surfperch dataset
- **Crab –** the site-wide EPC calculated as the UCL of the entire EW crab dataset
- **Sediment –** the concentration in the individual and composite sediment samples collected closest to the mussel location

For each location, the dietary COPC concentration will be calculated using the EPC for each prey item and sediment in Equation 4-2. The uncertainties in using mussel data as a surrogate for shrimp, including the use of mussel data rather than benthic invertebrate data as a surrogate, will be evaluated in the uncertainty section. The discussion of uncertainties will focus on the appropriateness of using mussel data as a surrogate for shrimp instead of data from benthic invertebrate samples, and whether mussels appear to bioaccumulate most COIs to a similar degree as shrimp.

4.1.3 Water approach

The water approach will be used to estimate exposure to fish through direct water contact for water COPCs. This section presents the water exposure assumptions and exposure calculation methods.

The water exposure approach requires an approximation of the COPC concentration in the water ventilated by the ROC. All fish ROCs will be assumed to be equally exposed to surface water throughout the EW. However, as discussed below, because rockfish have small home-ranges, a small-scale exposure analysis for rockfish will also be conducted.

Both new and existing water data will be used to calculate EPCs. Prior to calculating EPCs, a usability analysis of existing water data will be conducted to determine appropriate methods for combining the datasets. The site-wide EPC for each COPC will be calculated as the 95% UCL on the mean using all data in the combined dataset. ProUCL will be used to calculate the UCL as described in Section 3.2.1.

Because individual brown rockfish foraging ranges are smaller than the EW, there is uncertainty as to whether site-wide exposures are reflective of exposures to a subset of the EW rockfish population. To assess exposure of these subpopulations on a smaller scale, COPC concentrations in water samples at each water sampling location will also be compared to TRVs. The results of this evaluation will be provided to EPA prior to the submittal of the draft ERA, and the results will be used to finalize the approach for identifying COCs for water for subpopulations of rockfish.

4.2 EFFECTS ASSESSMENT

The effects assessment will present NOAEL and LOAEL TRVs for each COPC for the tissue residue and dietary risk analysis approaches. For the water approach, WQS will be used. If no WQS are available, national AWQC will be used. Criteria will be evaluated to ensure that they are based on survival, growth, and reproduction of invertebrates and fish. Criteria based on other receptors and endpoints will not be used. Criteria based on dissolved concentrations will be used when available. If no standards or criteria are available, or if criteria are unacceptable, then water-based NOAEL and LOAEL TRVs will be selected following the same methods used for selecting fish tissue and dietary TRVs. TRVs will be the same for all fish, except that TRVs based on reproduction will not be applied to juvenile Chinook salmon. Only growth and survival endpoints will be evaluated for assessing risks to juvenile Chinook salmon because egg and embryo life stages do not occur in the EW and because their exposure in the EW as adults is limited.[11] In cases where the fish TRV is based on reproduction, a unique TRV will be selected for juvenile Chinook salmon. The TRVs selected for the effects analysis are the same as those used in the COPC selection process; the methods for deriving these values are described in Section 2.2.2. The effects assessment will summarize all of the acceptable studies considered in deriving the TRV for each COPC and will discuss the rationale for selecting the NOAEL and LOAEL TRVs.

[11] If acceptable reproduction TRVs are identified in which fish were exposed solely as juveniles, they will be considered for juvenile Chinook salmon.

4.3 PREDICTED RISKS

4.3.1 Site-wide evaluation

Risk will be estimated for each ROC-COPC pair by calculating site-wide hazard quotients (HQs) from respective EPCs or dietary COPC concentrations and TRVs for each of the three risk analysis approaches (tissue residue, dietary, and water) as described below.

- **Tissue-residue approach –** HQs will be calculated for the metals, mercury, TBT, and selenium and all organic chemicals (excepting PAHs) as the quotient of the site-wide tissue-residue EPC divided by the selected tissue-residue NOAEL and LOAEL TRVs.
- **Dietary approach –** HQs will be calculated for PAHs and all metals (except mercury, selenium and TBT) as the quotient of the site-wide dietary COPC concentration divided by the selected dietary NOAEL and LOAEL TRVs.
- **Water approach –** HQs will be calculated for water COPCs as the quotient of the site-wide water EPC divided by the selected water TRV.

For English sole and rockfish tissue COCs will be identified based on tissue residue concentrations above the LOAEL TRV and dietary COCs will be identified based on dietary concentrations above the dietary LOAEL TRVs. Water COCs will be identified based on HQs greater than or equal to 1 based on the site-wide water EPC.

4.3.2 Location-specific evaluation

In addition to site-wide HQs, location-specific tissue, dietary, and water HQs will be presented for rockfish to evaluate the potential for adverse effects at a smaller than site-wide scale. The same methods used for site-wide HQ calculations will be used for this analysis, except that location-specific exposure data will be used. The following location specific HQs will be calculated:

- **Tissue-residue approach –** HQs will be calculated for each rockfish collection location using the COPC concentrations in individual rockfish rather than the site-wide EPCs.
- **Dietary approach –** HQs will be calculated for each mussel sample and benthic invertebrate tissue location using the dietary COPC concentrations calculated for each location rather than the site-wide dietary COPC concentrations.
- **Water approach –** HQs will be calculated for each water sample location using COPC concentrations in water samples from a location rather than the site-wide EPCs.

The location specific HQs calculated for rockfish will be used to identify COCs for rockfish. Tissue, dietary and water concentrations greater than or equal to the LOAEL TRVs will result in the identification of COCs for rockfish.

4.3.3 Evaluation of threatened and endangered species

For threatened or endangered species, risks to individuals are evaluated (EPA 1998), although specific guidance regarding this approach is not available. At other EPA Region 10 Superfund sites, such as Coeur d'Alene, Blackbird Mine, Portland Harbor, and LDW, greater emphasis has been placed on the NOAELs than on the LOAELs for the protection of threatened or endangered species. Thus, for all ROCs, except juvenile Chinook salmon (which is a federally threatened species), the primary emphasis of the risk characterization will be based on LOAEL HQs, whereas, for juvenile Chinook salmon, the primary emphasis of the risk characterization will be based on the NOAEL HQs. The HQs calculated for juvenile Chinook will be used to identify COCs for juvenile Chinook. Tissue, dietary and water concentrations greater than or equal to the NOAEL TRVs will result in the identification of COCs for juvenile Chinook.

4.3.4 Weight-of-evidence approach

Risk conclusions will be made for each line of evidence for the assessment of risk to the fish receptors and the lines of evidence will be used collectively, to draw overall conclusions that can inform risk management decisions for the site. Chemicals contributing to risk will be identified based on the:

- Magnitude of HQ
- Spatial and temporal distribution
- Degree of uncertainty regarding exposure and effects

A weight-of-evidence approach will be applied if there is disagreement among multiple lines of evidence. In the case of the fish receptors there may be disagreement between the assessments of dietary exposure, and the assessment based on surface water concentrations and tissue residues. Additionally, the rockfish will be assessed on both a site-wide basis as well as a location-specific basis. In these instances the uncertainties associated with both the assessment of exposure and the assessment of effects will be examined in order to identify the lines of evidence with the greatest degree of certainty. Both the strength of the exposure data and the uncertainties associated with effects data would be evaluated. In the risk conclusion section of the fish risk characterization, results will be presented in a table that summarizes risk for each ROC/COPC pair. Uncertainties in risk conclusions for fish will be presented in the risk characterization section, as well as in a separate uncertainty section that summarizes uncertainties for the entire risk assessment.

5 Wildlife Risk Approach

This section describes each of the three components for the wildlife risk assessment: the exposure assessment, the effects assessment, and risk characterization. The overall approach to the wildlife risk assessment is designed to address the assessment endpoints identified for pigeon guillemot, osprey, river otter, and harbor seal in Table 2-1. Specifically, the following risk questions will be answered by conducting the ERA:

- Are concentrations of COPCs in the diet of birds that forage in the EW greater than concentrations reported in the literature to cause reduced survival, growth, or reproduction in birds?
- Are concentrations of COPCs in the diet of mammals that forage in the EW greater than concentrations reported in the literature to cause reduced survival, growth, or reproduction in mammals?

5.1 Exposure Assessment

The exposure of wildlife will be evaluated through the dietary pathway. Direct contact pathways were also identified in the CSM and were determined to be insignificant relative to the dietary exposure pathway.[12] The approach for estimating the chemical doses in the diet along with the exposure factors that will be used for each ROC are described below.

5.1.1 Daily dose estimate

Estimates of the daily ingested dose of each chemical for each receptor will be calculated for ingestion of prey in the diet, as well as for incidental ingestion of water and sediment. The daily doses will be estimated using the following equation:

$$\text{Daily Dose} = \frac{\left[\left(IR_{diet} \times C_{diet}\right) + \left(IR_{water} \times C_{water}\right) + \left(IR_{sed} \times C_{sed}\right)\right] \times SUF}{BW} \qquad \textbf{Equation 5-1}$$

Where:

Daily Dose	=	COPCs ingested per day via diet, water, and sediment (mg/kg bw/day)
IR_{diet}	=	dietary ingestion rate (kg/day dw)
C_{diet}	=	concentration in diet (mg/kg dw)
IR_{water}	=	water ingestion rate (L/day)

[12] Direct (or dermal) contact with sediment was considered a complete exposure pathway for pigeon guillemot, river otter, and harbor seal. However, risks from sediment contact are considered to be insignificant relative to those from ingestion (EPA 2000b). Direct contact with water was also considered a complete exposure pathway for all four wildlife ROCs but was assumed to be insignificant because feathers on birds and fur on mammals limit direct contact of skin with contaminated media.

C_{water} = concentration in water (mg/L)
IR_{sed} = sediment ingestion rate (kg/day dw)
C_{sed} = concentration in sediment (mg/kg dw)
SUF = site use factor (unitless); fraction of time that a receptor spends foraging in the EW relative to the entire home range
BW = ROC body weight (kg ww)

The chemical concentration in the diet will be calculated from concentrations in each prey component of the ROC diet and estimates of each prey component's fraction of the total diet, as follows:

$$C_{diet} = \sum_{i=1}^{n} X_i C_i$$ **Equation 5-2**

Where:

C_{diet} = COPC EPC in the diet (mg/kg dw)
X_i = proportion of a particular prey item in the diet (unitless)
C_i = COPC concentration in the prey item (mg/kg dw)
n = number of dietary items

The dietary fraction of each component in each ROC's diet that will be used in the risk assessment is based on information from the literature. The dietary fractions to be used for each ROC and the assumptions used to derive them are described in detail in Section 5.1.2.

Site use factors, body weights, and dietary, water, and sediment ingestion rates (IRs) vary among ROCs. The body weights and dietary, water, and sediment IRs that will be used in the ERA have been obtained from the literature for each receptor. Incidental sediment IRs have been calculated as a specified percentage of the food IR for each ROC. As shown in Equation 5-1, ingestion of prey alone represents 100% of the dietary IR, with sediment making up an additional component of the diet. All COPCs will be conservatively assumed to have the same bioavailability in the field as in the laboratory toxicity study that provides the basis for the TRV in all media.

For dioxins, furans, and dioxin-like PCBs, dietary risks to birds and mammals will be evaluated using a TEQ approach. Using this approach, the potencies of specific individual dioxin and furan congeners and dioxin-like PCB congeners relative to that of 2,3,7,8-TCDD are quantified using TEFs. The TEFs that will be used for birds and mammals are those developed by the World Health Organization (WHO); bird values are from 1998, and mammalian values are from 2006 (Van den Berg et al. 1998; 2006), as described in Section 4.1.1. When the dioxin, furan, or PCB congener concentrations are reported as undetected, then the TEF will be multiplied by half the reporting limit.

5.1.2 Exposure factors

This section presents the exposure factors used in Equations 5-1 and 5-2 to calculate the daily exposure dose for each ROC, including dietary fractions of prey items, ingestion rates of prey, water, and incidental sediment, site use factors, and body weights. Tables 5-1 and 5-2 summarize these values, and the following subsections provide details of exposure factor assumptions and sources of information for each ROC. The ingestion rates and body weights are the same as those used in the LDW ERA for the three of the four EW ROCs that were included in the LDW ERA (i.e., osprey, river otter, and harbor seal).

Table 5-1. Dietary fractions of prey items used in wildlife exposure calculations

ROC	Fraction of Prey Item in Diet (by weight)							
	Shiner Surfperch	English Sole	Juvenile Chinook Salmon	Brown Rockfish	Crabs	Shrimp	Clams	Mussels
Pigeon guillemot	0.143	0.143	0.143	0.143	0.143	0.143	0	0.143
Osprey	0.50	0	0.50	0	0	0	0	0
River otter	0.22	0.22	0.22	0.22	0.10	0	0.01	0.01
Harbor seal	0.25	0.25	0.25	0.25	0	0	0	0

ROC – receptor of concern

Table 5-2. Exposure factor values for wildlife ROCs

ROC	Gender[a]	Body Weight (kg ww)	Dietary Ingestion Rate (kg/day dw)	Incidental Water Ingestion Rate (kg/day)	Incidental Sediment Ingestion Rate (kg/day dw)	Site Use Factor (unitless)[b]
Pigeon guillemot	male	0.483	0.023	0.036	0.00046	0.5
	female	0.530	0.025	0.039	0.00050	
	average	0.485	0.023	0.036	0.00046	
Osprey	male	1.5	0.076	0.077	0.00076	0.41
	female	1.8	0.091	0.087	0.00091	
	average	1.7	0.083	0.083	0.00083	
River otter	male	9.2	0.30	0.73	0.0060	0.23
	female	7.9	0.26	0.64	0.0052	
	average	8.6	0.28	0.68	0.0056	
Harbor seal	male	85	0.62	5.4	0.012	0.1
	female	77	0.58	4.9	0.012	
	average	81	0.60	5.1	0.0012	

a Female values will be used for COPCs with a TRV based on a reproductive endpoint, and average values will be used for COPCs with a TRV based on a growth or survival endpoint. Average values will be used for the COPC screen (Section 2.2.3).

b Site use factor is the fraction of a receptor's total foraging time that is spent in the EW.

COPC – chemical of potential concern
dw – dry weight
EW – East Waterway
ROC – receptor of concern
TRV – toxicity reference value
ww – wet weight

5.1.2.1 Pigeon guillemot

Body Weight

Female pigeon guillemots weighed in April in California were slightly heavier than males (average of 487 and 483 g, respectively), but females weighed during egg laying (May to June) in Alaska were substantially heavier than males (530 and 462 g, respectively) (Ewins 1993). In the EW ERA, female body weight values will be used for evaluating risk for COPCs with TRVs based on a reproductive endpoint for all bird and mammal species. Therefore, the female body weight during the reproductive season will be used for pigeon guillemots. Average body weights will be used for the remaining COPCs with growth or survival endpoints for all bird and mammal species, so the average of the male and female pigeon guillemots weighed in April prior to breeding (485 g) will be used for pigeon guillemots. Because of the uncertainty in these body weights, the effect of using different weights on the HQ calculations will be evaluated in the uncertainty section.

Dietary Ingestion Rate

The dietary IR in wet weight has been estimated as 20% of the body weight (Ewins 1993). Dietary IR values will be converted from wet weight to dry weight using the average moisture content in fish collected from the EW; for the purpose of this technical memorandum, preliminary dry weight IRs have been calculated using the average moisture content in LDW fish of 76%. In the ERA, IRs will be calculated using the average moisture content in EW fish. Dietary IR values of 0.023 and 0.025 kg dw/day have been calculated for male and female pigeon guillemots, respectively. An average IR value of 0.023 kg dw/day has been calculated.

Water Ingestion Rate

The water IR has been estimated as a function of the pigeon guillemot's body weight, using an allometric equation recommended in *Wildlife Exposure Factors Handbook* (EPA 1993), as follows:

$$IR_{water} = 0.059 \times BW^{0.67}$$ **Equation 5-3**

Where:

IR_{water} = water ingestion rate (L/day)
BW = body weight (kg)

The calculated male and female water IR values are 0.039 and 0.036 L/day, respectively. The average water IR value calculated is 0.036 L/day.

Incidental Sediment Ingestion Rate

Information on rates of incidental sediment ingestion by pigeon guillemots was not available. Pigeon guillemots may ingest a small amount of sediment while foraging for food. Thus, sediment ingestion has been estimated to be 2% of the pigeon guillemot's diet.

Composition of Diet

Pigeon guillemots are known to feed on bottom-dwelling organisms in water up to 45 m deep (Ewins 1993). It is estimated that the optimal diving and foraging efficiency for pigeon guillemots is in water 10 to 20 m deep (Ewins 1993). Most foraging is conducted in benthic habitats, although pigeon guillemots also feed in the water column (Ewins 1993). Because no data have been found to indicate that pigeon guillemots would not also feed in shallower water, both subtidal and intertidal areas of the EW will be considered foraging habitat.

Pigeon guillemots are "generalists" have been known to feed on over 50 species of benthic fish and invertebrates (Kuletz 1998). Fish and invertebrates in pigeon guillemot diets from Alaska, Oregon, and British Columbia include sand eels, Pacific sandfish, capelin, cods, sculpins, gunnels, blennies, gadids, Pacific sand lance, prickleback, flatfish, Pacific herring, crabs, shrimps, with occasional polychaetes, gastropods, and bivalve mollusks (Ewins 1993; Golet et al. 2000; Kuletz 1998; Litzow et al. 2000). There may be considerable variation in diet as a function of habitat and year, as well as among individuals and pairs (Ewins 1993; Kuletz 1998; Litzow et al. 2000). Because of the wide variety of species that may be consumed, it is assumed that pigeon guillemots foraging in the EW consume equal proportions of shiner surfperch, English sole, juvenile Chinook salmon, brown rockfish, crabs, mussels, and shrimp.

Site Use

Pigeon guillemots are present in the Puget Sound region year-round (Seattle Audubon Society 2008), and their nests have been observed under the Terminal 18 (T-18) piers (Hotchkiss 2007). Limited data are available to estimate the foraging range of pigeon guillemots. Litzow and Piatt (2003) observed that radio-tagged pigeon guillemots foraged only in the area in which they nested, although the size of that area was not defined. Data summarized by Ewins (1993) indicates that most breeders feed within 7 km of their nests. Based on this limited information, it is assumed that pigeon guillemots nesting along the EW obtain approximately half of their diet from within the EW site. Thus, the proposed site use factor for pigeon guillemot is 0.5.

5.1.2.2 Osprey

Body Weight

Representative body weights for adult male and female osprey (1.5 and 1.8 kg, respectively) have been obtained from Poole (1989), as cited in Poole et al. (2002). The average of the male and female body weights was 1.7 kg.

Dietary Ingestion Rate

The dietary IR in wet weight has been estimated as 21% of the body weight (Poole 1983; as cited in EPA 1993). Dietary IR values will be converted from wet weight to dry weight using the average moisture content in fish collected from the EW; for the purpose of this technical memorandum, preliminary dry weight IRs have been calculated using the average moisture content in LDW fish of 76%. In the ERA, IRs will be calculated using the average moisture content in EW fish. Dietary IR values of 0.076 and 0.091 kg dw/day have been calculated for male and female osprey, respectively. An average dietary IR value of 0.083 kg dw/day has been calculated.

Water Ingestion Rate

The water IR has been estimated as a function of the osprey's body weight, using the allometric equation recommended in *Wildlife Exposure Factors Handbook* (EPA 1993). This equation is presented in Section 5.1.2.1 (Equation 5-3). Water IR values of 0.077 and 0.087 L/day have been calculated for male and female osprey, respectively. The average water IR value calculated is 0.083 L/day.

Incidental Sediment Ingestion Rate

Data on incidental sediment ingestion by osprey were not available. Osprey may ingest a small amount of sediment while foraging for fish in shallow intertidal water. Because osprey generally catch fish from only the top 1 m of the water surface, they are not expected to contact sediment in deeper subtidal areas. Thus, sediment ingestion has been estimated to be 1% of the osprey's diet, and it has been assumed that only intertidal sediment would be ingested.

Composition of Diet

Osprey feed almost exclusively on live fish; at least 99% of their prey items are live fish in most published accounts (Poole et al. 2002). Ospreys can penetrate about 1 m below the water surface. Therefore, they generally catch pelagic fish or those that frequent shallow flats and shorelines. Ospreys may infrequently ingest other types of vertebrate prey, such as birds, reptiles, and small mammals. A west-central Idaho osprey study reported 89% of fish ingested by osprey were 11 to 30 cm long, suggesting a preference for medium-sized fish (Van Daele and Van Daele 1982). During a US Geological Survey (USGS) study in 2006, osprey were observed while returning with prey to two nests along the EW (Davis 2007).[13] Diets at the T-18 nest consisted of 15.4% salmonids, 76.9% freshwater fish, and 7.7% unknown fish, and diets of birds frequenting the Terminal 104 (T-104) nest consisted of 50% surfperch and 50% salmonids. Based on these data, it will be assumed that the osprey diets consist of the maximum percentage of EW fish found among both nests: 50% surfperch and 50% salmonids.

Site Use

Two osprey nest boxes are located along the EW: one at T-104 and one at T-18 (Blomberg 2007). In 2006, Washington State Department of Fish and Wildlife (WDFW) reported 10 osprey nest sites located along the Duwamish River, which included nests along the EW (Thompson 2006). The distance osprey travel from their nests to forage depends on the availability of fish near the nest (Van Daele and Van Daele 1982). Preliminary USGS data were available on foraging locations for the two EW osprey nests in 2006 (Davis 2007). For the nest at T-18, prey fish were captured from Lake Washington, the LDW, and Elliott Bay (76.9, 15.4, and 7.7%, respectively); and for the nest at T-104, prey fish were captured from the LDW and Puget Sound (66.7 and 33.3%, respectively). Marine species identified included salmonids and surf perch. To calculate the site use factor, it has been assumed that all LDW fish from each nest were captured from the EW. The average percentage of EW fish assumed to be captured for both nests is 41%; thus, the proposed site use factor for osprey is 0.41.

5.1.2.3 River otter

Body Weight

Adult body weights of 9.2 and 7.9 kg have been assumed for male and female river otter, respectively based on a study by Melquist and Hornocker (1983), as cited in EPA (1993). The average of the male and female body weights is 8.6 kg.

Dietary Ingestion Rate

The dietary IR has been estimated as a function of the metabolic rate and the caloric content of the prey using the following equation:

$$IR_{diet} = \frac{FMR}{ME} \times \frac{0.001\,\text{kg food}}{\text{g food}}$$ **Equation 5-4**

Where:

IR_{diet} = food ingestion rate (kg/day dw)
FMR = free-living metabolic rate (kilocalories [kcal]/day)
ME = average metabolizable energy of the total diet (kcal/g dw)

[13] Additional dietary data for osprey nesting along the EW may be available from USGS or USFWS for inclusion in the ERA.

The FMRs for males and females have been calculated to be 1,340 and 1,180 and kcal/day, using an equation developed by Nagy (1987), as cited in EPA (1993), for placental mammals:

$$FMR\ (kcal/day) = 0.800 \times BW^{0.813}$$ **Equation 5-5**

where body weight is expressed in grams. The ME value used for mammals on a diet of fish is that calculated by Nagy (1987), as cited in EPA (1993) (4.47 kcal/g dw). The calculated dietary IRs for males and females were 0.30 and 0.26 kg dw/day, respectively (Table 5-2). An average dietary IR value of 0.28 kg dw/day was calculated.

Water Ingestion Rate

The water IR has been estimated as a function of the river otter's body weight, using an allometric equation recommended in EPA (1993). This equation was developed by Calder and Braun (1983), as cited in EPA (1993):

$$IR_{water} = 0.099 \times BW^{0.90}$$ **Equation 5-6**

Where:

IR_{water} = water ingestion rate (L water/day)
BW = body weight (kg)

Water IR values for males and females (0.73 and 0.64 L/day, respectively) were calculated. The average water IR value calculated was 0.68 L/day.

Incidental Sediment Ingestion Rate

Data were not available to estimate the amount of sediment ingested incidentally by river otters. A small amount of sediment could be ingested when river otters forage on crabs and benthic fish species; therefore, the incidental sediment IR has been estimated to be 2% of the dietary IR. It has been assumed that river otters incidentally ingest sediment from both intertidal and subtidal areas of the EW.

Composition of Diet

River otters are opportunistic carnivores that take advantage of food that is most abundant and easiest to catch. Fish are their primary prey (Kurta 1995; Larsen 1984; Stenson et al. 1984; Wise et al. 1981). River otters catch fish by diving and ambushing or chasing, and obtain invertebrates by digging in the substrate (Coulter et al. 1984). Slower-moving fish, such as suckers, carp, chubs, and bullheads, are generally eaten most frequently (Kurta 1995; Wise et al. 1981). Studies in coastal southeast Alaska and British Columbia found that river otters feed primarily on sculpin, surfperch, and flatfish, with greenling, salmon, and rockfish making up lesser portions of the diet (Larsen 1984; Stenson et al. 1984). Other components of the river otter diet include aquatic invertebrates (including crayfish, mussels, clams, and aquatic insects), frogs, snakes, and occasionally mammals and birds (Coulter et al. 1984).

River otters generally ingest fish ranging from 7.6 to 41 cm in length (Gilbert and Nancekivell 1982; Greer 1955; both as cited in EPA 1993); although Toweill (1974) found that many of the salmon preyed upon by river otters in western Oregon were up to an estimated 80 cm in length. These salmon were taken in coastal streams where fish enter the rivers to spawn. Local river otters feed largely on fish but will also feed on crabs and sometimes mussels and clams (Strand 1999).

The proportion of prey types ingested by river otter for this assessment has been based on Larsen's (1984) study of river otters in southeastern Alaska. This study was used because it was the only study from the Pacific Northwest that reported remains in scat on a volume basis rather than as a frequency of occurrence. Larsen (1984) reported the following proportions of prey ingested by river otters: 86% fish, 10% crabs, 2% invertebrates other than crabs, 1% birds, and 1% mammals and plant material. Thus, for this assessment, it is assumed that river otters ingest 88% fish, 10% crabs, and 1% each of mussels and clams. Based on feeding habits of river otters documented in coastal southeast Alaska and British Columbia (Larsen 1984; Stenson et al. 1984), any of the four types of fish tissue for which chemistry data were available in the EW might be ingested. Because no site-specific information was available on fish preference of river otters, it has been assumed that shiner surfperch, English sole, juvenile Chinook salmon, and brown rockfish are ingested in equal proportions of the 88% of the river otter's diet that is fish.

Site Use

Anecdotal information indicates that a river otter family lives year-round on Kellogg Island in the Duwamish River, although otters were not observed during wildlife surveys by Cordell (2001). River otters are almost exclusively aquatic and prefer food-rich habitats such as the lower portions of streams and rivers, estuaries, and lakes and tributaries that feed rivers (Tabor and Wight 1977; Mowbray et al. 1979). In streams, the river otter's home range can average 30 km (19 mi) (Melquist and Hornocker 1983). At any given time, river otters generally occupy only a few kilometers of stream but often move from one area to another (Nebraska Game and Parks Commission 2000). A radio-tracking study of relocated river otters was conducted as part of the New York State Department of Environmental Conservation river otter reintroduction program. This study showed that river otter ranges were from 1.5 to 22.4 km long, with an average length of 10 km (6 mi) for individuals monitored in western New York State (Spinola et al. 1999; as cited in EPA 2000c).

No studies that document usage of the EW by river otters were found. However, it is assumed that river otters that may potentially inhabit Kellogg Island have a home range of 10 km, as documented in Spinola et al. (1999; as cited in EPA 2000c). Assuming river otters forage in the areas 5 km north and south of Kellogg Island equally and because the EW is approximately 2.3 km long (23% of 10 km), the proposed site use factor for river otters is 0.23.

5.1.2.4 Harbor seal

Body Weight

Body weights for adult male and female harbor seals (84.6 and 76.5 kg, respectively) have been based on a study by Pitcher and Calkins (1979), as cited in the *Wildlife Exposure Factors Handbook* (EPA 1993). The average of the male and female body weights was 80.6 kg.

Dietary Ingestion Rate

The dietary IR for harbor seals has been calculated using an allometric equation developed by Boulva and McLaren (1979) for harbor seals from eastern Canada, as cited in the *Wildlife Exposure Factors Handbook* (EPA 1993):

$$IR_{diet} = 0.089 \times BW^{0.76}$$ **Equation 5-7**

Where:

IR_{diet} = dietary ingestion rate (kg/day ww)

BW = harbor seal body weight (kg)

The calculated wet weight dietary IR values will be converted to dry weight using the average moisture content in whole-body fish from the EW; for the purposes of this technical memorandum, approximate dry weight IRs have been calculated using the average moisture content in LDW fish of 76%. In the ERA, IRs will be calculated using the average moisture content in EW fish. Dietary IRs calculated for males and females were 0.62 and 0.58 kg dw/day, respectively (Table 5-2). An average diet IR value of 0.60 kg dw/day was calculated.

Water Ingestion Rate

The water IR has been estimated as a function of the harbor seal's body weight, using the allometric equation recommended in EPA (1993). This equation is presented in Section A.5.1.2.3 (Equation 5-6). Using the male and female body weights of the harbor seal, the calculated water IR values were 4.9 and 5.4 L/day, respectively. The average water IR value calculated was 5.1 L/day.

Incidental Sediment Ingestion Rate

Data on incidental sediment ingestion by harbor seals were not available, but it is possible that a small amount of sediment could be incidentally ingested while foraging on bottom fish. Therefore, the sediment IR has been assumed to be 2% of the dietary IR. It has been assumed that harbor seals ingest sediment from both intertidal and subtidal areas of the EW.

Composition of Diet

Harbor seals are opportunistic feeders, selecting prey based on availability and ease of capture (Pitcher and Calkins 1979; Pitcher 1980; Schaffer 1989). Their diet can vary seasonally with local abundance and includes bottom-dwelling fishes, invertebrates such as octopus and squid, and species that congregate for spawning (Pitcher and Calkins 1979; Everitt et al. 1981; Lowry and Frost 1981; Roffe and Mate 1984). In Washington, the most important prey include Pacific whiting, tomcod, walleye pollock, flatfish, Pacific herring, shiner surfperch, plainfin midshipman, and sculpin (NMFS 1997). Fish ingested were generally between 4 and 28 cm in length (Brown and Mate 1983). Harbor seals may also prey on salmon during upriver spawning migrations of adults or downriver outmigrations of juveniles, although site-specific data were not available on the dietary importance of migrating salmon to local seal populations. Because site-specific information was not available on the amount of each type of fish ingested, it is assumed that juvenile Chinook salmon, English sole, shiner surfperch, and brown rockfish are ingested in equal proportions.

Site Use Factor

Harbor seals are commonly seen in Elliott Bay and occasionally enter the EW (Kenney 1982). Harbor seals have been shown to forage over large distances, ranging from 5 km (3 mi) (Stewart et al. 1989) to 55 km (34 mi) (Beach et al. 1985). In Puget Sound, harbor seals generally forage within 8 to 13 km (5 to 8 mi) of their haulout areas established as pupping sites (Jeffries 2001). The closest known pupping site to the EW is located at Blakely Rocks off the southeast end of Bainbridge Island, approximately 12 km (7 mi) from the EW. Site-specific information on harbor seal usage of the EW is limited. The WDFW observed harbor seals infrequently in the EW during an intensive survey conducted from December 1998 to June 1999, which monitored the EW, West Waterway, and LDW up to the 16th Avenue South Bridge for the presence of sea lions and seals for a total of 307 hours on 52 days (Walker 1999). The EW was monitored for a total of 28.25 hours on 29 days; one harbor seal was observed during this time. While harbor seals have been observed in Elliott Bay and may use log booms to haul out, they are not known to aggregate in large numbers (Jeffries 2001). The EW may be a preferential feeding area during salmonid outmigration from March through August. For example, in the Columbia River, salmonids appear to be targeted as prey by seals in the spring and fall when they are abundant and available in the river (NMFS 1997).

Data from the WDFW survey (Walker 1999) were used to establish a site use factor for risk calculations. The following conservative assumptions were used for the one harbor seal observed in the EW: 1) it has been assumed that the seal obtained all of its food from the EW on that day; and 2) site usage from December through June accurately represents usage during other times of the year. Based on these assumptions, the site use factor is equal to 1/29 or 0.03. For the purposes of the ERA a site use factor of 0.1 will be used for the seal in order to account for potentially higher use of the site by seals during periods of salmon migration.

5.1.3 Exposure concentrations

For all ROC-COPC pairs established in the COPC selection process (Section 2.3.3), a single site-wide EPC will be calculated as the UCL calculated over all samples for each particular prey species in an ROC's diet (i.e., the C_i component in Equation 5-2). One exception will be shrimp, which will be represented by a single sample. For sediment, UCLs will be calculated using all samples for evaluation of risk to pigeon guillemot, otter, and harbor seal, and will be calculated using only intertidal samples for evaluation of risk to osprey. ProUCL will be used to calculate the UCL as described in Section 3.2.1.

5.2 Effects Assessment

The effects assessment will present NOAEL and LOAEL TRVs for each COPC. The methods for deriving these values are described in Section 2.3.2. The effects assessment will summarize all of the acceptable studies considered in deriving the TRV for each COPC and will discuss the rationale for selecting the NOAEL and LOAEL TRVs.

5.3 Risk Characterization

Risk will be evaluated for each ROC-COPC pair by comparing the dietary dose to both the NOAEL and LOAEL TRVs. HQs will be calculated as the dietary dose divided by the TRV. HQs greater than 1.0 indicate that the exposures are estimated to be greater than toxicological benchmarks and will be used to identify COCs for wildlife. Such a finding is generally regarded as indicating a potential for adverse effects, particularly if the benchmark is an effects concentration (or dose) at which adverse effects were observed (i.e., a LOAEL). HQs may also be calculated based on a NOAEL. The potential for adverse effects associated with a NOAEL HQ greater than 1.0 is uncertain unless the LOAEL is also assessed because the true threshold for effects occurs at a concentration (or dose) somewhere between the NOAEL and LOAEL. An exposure greater than the NOAEL and less than the LOAEL may or may not result in any adverse effect. In the risk conclusion section of the wildlife risk characterization, results will be presented in a table that summarizes risk for each ROC/COPC pair. Uncertainties in risk conclusions for wildlife will be presented in the risk characterization section, as well as in a separate uncertainty section that summarizes uncertainties for the entire risk assessment.

6 Uncertainty

Uncertainties will be evaluated for all steps of the risk assessment and will be presented as they occur in the risk assessment process. The discussion of uncertainties in the problem formulation will focus on the selection of ROCs, assessment endpoints, and exposure pathways. The discussion of uncertainties in the exposure assessment will focus on the availability or relevance of site-specific data to estimate or measure exposure, as well as any parameters used in estimating exposure. The discussion of uncertainties in the effects assessment will focus on the availability and relevance of toxicological data, the majority of which have been selected from regulatory guidelines and the literature, except for the site-specific toxicity test results for benthic invertebrates. A sensitivity analysis of the risk estimates will also be conducted to identify key uncertainties in the exposure estimates (i.e., to identify those parameters with a strong influence on risk conclusions). For each type of uncertainty, the analysis will address whether the uncertainty would lead to an overestimate or underestimate of risk, or if it is not possible to determine if risks would overestimated or underestimated. Also, where possible, the relative degree of uncertainty will be discussed. A summary of uncertainties and their effect on risk conclusions or risk management decisions will be provided as part of the risk conclusions section of the ERA.

7 References

Alden RW, III, Weisberg SB, Ranasinghe JA, Dauer DM. 1997. Optimizing temporal sampling strategies for benthic environmental monitoring programs. Mar Poll Bull 34(11):913-922.

Anchor, Windward. 2005. East Waterway Operable Unit, Phase 1 Removal Action completion report. Anchor Environmental L.L.C. and Windward Environmental LLC, Seattle, WA.

Anchor, Windward. 2008. East Waterway Operable Unit: Existing information summary report. Anchor Environmental, L.L.C., and Windward Environmental LLC, Seattle, WA.

Anchor, Windward, Battelle, KPFF, Coast and Harbor Engineering. 2008. Final conceptual site model and data gaps analysis report. East Waterway Operable Unit supplemental remedial investigation/feasibility study. Prepared for Port of Seattle. Anchor Environmental, L.L.C., Seattle, WA; Windward Environmental LLC, Seattle, WA; Battelle Memorial Institute, Seattle, WA; KPFF Consulting Engineers, Seattle, WA; and Coast and Harbor Engineering, Edmonds, WA.

Barrick R, Becker S, Brown L, Beller H, Pastorok R. 1988. Sediment quality values refinement: Volume I. 1988 update and evaluation of Puget Sound AET. Prepared for Puget Sound Estuary Program (PSEP), US Environmental Protection Agency, Region 10. PTI Environmental Services, Inc., Bellevue, WA.

Battelle. 1996. Final report for the PCB Aroclor and congener analyses on fish tissue samples from the Elliott Bay/Duwamish River project. Pacific Northwest Division, Battelle Marine Research Laboratory, Sequim, WA.

Beach RA, Geiger C, Jeffries SJ, Treacy SD, Troutman BL. 1985. Marine mammals and their interactions with fisheries of the Columbia River and adjacent waters, 1980-1982. Third annual report. Wildlife Management Division, Washington Department of Wildlife, Olympia, WA.

Blomberg G. 2007. Personal communication (telephone conversation with Susan McGroddy, Windward Environmental, regarding osprey nests along the East Waterway). Port of Seattle, Seattle, WA. July 25, 2007.

Borgmann U, Janssen C, Blust RJP, Brix KV, Dwyer RL, Erickson RJ, Hare L, Luoma SN, Paquin PR, Roberts CA, Wang W-X. 2005. Incorporation of dietborne metals exposure into regulatory frameworks. Chapter 5. In: Meyer JS, Adams WJ, Brix KV, Luoma SN, Mount DR, Stubblefield WA, Wood CM, eds, Toxicity of dietborne metals to aquatic organisms. Society of Environmental Toxicology and Chemistry (SETAC), Pensacola, FL, pp 153-190.

Boulva J, McLaren IA. 1979. Biology of the harbor seal, *Phoca vitulina*, in eastern Canada. Fish Res Board Can Bulletin 200:1-24.

Bragdon-Cook K. 1995. Chemical summing for LPAH, HPAH, benzofluoranthenes and total PCB's groups. Triennial review, Sediment Management Standards Rule, Chapter 173-204 WAC. Washington Department of Ecology, Olympia, WA.

Brown RF, Mate BR. 1983. Abundance, movements, and feeding habits of harbor seals, *Phoca vitulina*, at Nearts and Tillamook Bays, Oregon. USNMFS Fish Bull 81:291-301.

Bury NR, Walker PA, Glover CN. 2003. Nutritive metal uptake in teleost fish. J Exp Biol 206:11-23.

Calabrese EJ, Baldwin LA. 1993. Performing ecological risk assessments. Lewis Publishers, Chelsea, MI.

Calder WA, Braun EJ. 1983. Scaling of osmotic regulation in mammals and birds. Am J Physiol 244:R601-R606.

Clearwater SJ, Farag AM, Meyer JS. 2002. Bioavailability and toxicity of dietborne copper and zinc to fish. Comp Biochem Physiol Part C 132:269-313.

Cordell J. 2001. Personal communication (e-mail to Matt Luxon, Windward Environmental, Seattle, WA, regarding observations of juvenile chinook and other wildlife in the Lower Duwamish River). Researcher, Department of Fisheries, University of Washington, Seattle, WA. July 9.

Cordell JR, Tear LM, Jensen K, Luiting V. 1997. Duwamish river coastal America restoration and reference sites: Results from 1996 monitoring studies. Fisheries Research Institute, University of Washington, Seattle, WA.

Cordell JR, Tear LM, Jensen K, Higgins HH. 1999. Duwamish River coastal America restoration and reference sites: Results from 1997 monitoring studies. FRI-UW-9903. Fisheries Research Institute, University of Washington, Seattle, WA.

Cordell JR, Tear LM, Jensen K. 2001. Biological monitoring at Duwamish River coastal America restoration and reference sites: A seven-year retrospective. SAFS-UW-0108. Wetlands Ecosystem Team, School of Aquatic and Fisheries Sciences, University of Washington, Seattle, WA.

Coulter W, Adams CM, Decker DJ. 1984. River otter biology. In: New York's wildlife resources [online]. Department of Natural Resources, New York State College of Agriculture and Life Sciences, State University at Cornell University, Ithaca, NY. [Cited June 6, 2000.] Available from: http://www.nyotter.org/pages/biology.html.

Davis JW. 2007. Personal communication (e-mail to D. Williston, King County regarding osprey use of the East Waterway). Environmental Toxicologist, US Fish and Wildlife Service, Lacey, WA. December 13, 2007.

Ecology. 2008. Sediment sampling and analysis plan appendix: guidance on the development of sediment sampling and analysis plans meeting the requirements of the sediment management standards (chapter 173-204 WAC). Ecology Publication No. 03-09-043. Washington State Department of Ecology, Olympia, WA.

Edens FW, Garlich JD. 1983. Lead-induced egg production decrease in leghorn and Japanese quail hens. Poult Sci 62:1757-1763.

Edens FW, Benton E, Bursian SJ, Morgan GW. 1976. Effect of dietary lead on reproductive performance in Japanese quail, *Coturnix coturnix japonica*. Toxicol Appl Pharmacol 38:307-314.

EPA. 1992. Framework for ecological risk assessment. EPA/630/R-92/001. Risk Assessment Forum, US Environmental Protection Agency, Washington, DC.

EPA. 1993. Wildlife exposure factors handbook. EPA/600/R-93/187a. Office of Research and Development, US Environmental Protection Agency, Washington, DC.

EPA. 1997a. Ecological risk assessment guidance for Superfund: Process for designing and conducting ecological risk assessments. EPA/540/R-97/006. Interim final. Environmental Response Team, US Environmental Protection Agency, Edison, NJ.

EPA. 1997b. EPA Region 10 supplemental ecological risk assessment guidance for Superfund. EPA/910/R-97/005. Region 10 Office of Environmental Assessment Risk Evaluation Unit, US Environmental Protection Agency, Seattle, WA.

EPA. 1998. Guidelines for ecological risk assessment. EPA/630/R-95/002 F. Risk Assessment Forum, US Environmental Protection Agency, Washington, DC.

EPA. 2000a. Bioaccumulation testing and interpretation for the purpose of sediment quality assessment: status and needs. EPA-823-R-00-001. Bioaccumulation Analysis Workgroup, US Environmental Protection Agency, Washington, DC.

EPA. 2000b. Evaluation of dermal contact and inhalation exposure pathways for the purpose of setting Eco-SSLs. Draft ecological soil screening level guidance, Exhibit 1-3. US Environmental Protection Agency, Washington, DC.

EPA. 2000c. Hudson River PCBs reassessment. Revised baseline ecological risk assessment. US Environmental Protection Agency Region 2, New York, NY.

EPA. 2007a. Framework for metals risk assessment. EPA 120/R-07/001. Office of the Science Advisor, Risk Assessment Forum, US Environmental Protection Agency, Washington, DC.

EPA. 2007b. ProUCL Version 4.0. Statistical software for environmental applications for data sets with and without nondetect observations [online]. Technical Support Center for Monitoring and Site Characterization, US Environmental Protection Agency. Updated 6/14/07. Available from: http://www.epa.gov/nerlesd1/tsc/software.htm.

ESG. 1999. Waterway sediment operable unit, Harbor Island Superfund site. Assessing human health risks from the consumption of seafood: human health risk assessment report. Prepared for Port of Seattle, Todd Shipyards, and Lockheed-Martin for submittal to US Environmental Protection Agency, Region 10, Seattle, WA. Environmental Solutions Group, Inc., Seattle, WA.

Everitt RD, Gearin P, Skidmore JS. 1981. Prey items of harbor seals and California sea lions in Puget Sound, Washington. Murrelet 62: 83-8.

EVS. 1996a. Harbor Island sediment operable unit. Supplemental remedial investigation. Volume 1 of 2. EVS Consultants, Seattle, WA.

EVS. 1996b. Harbor Island sediment operable unit. Supplementary remedial investigation. Appendices, Vol 2 of 2. EVS Consultants, Seattle, WA.

Ewins PJ. 1993. Pigeon guillemot (*Cepphus columba*). In: Poole A, ed, The birds of North America online. Cornell Laboratory of Ornithology, Ithaca, NY.

Fresh KL, Rabin D, Simenstad CA, Salo EO, Garrison K, Matheson L. 1979. Fish ecology studies in the Nisqually Reach area of southern Puget Sound, Washington. FRI-UW-7904. Prepared for Weyerhauser Company. Fisheries Research Institute, University of Washington, Seattle, WA.

Frontier Geosciences. 1996. Mercury results in 18 fish samples for the Elliott Bay/Duwamish River project. Frontier Geosciences, Seattle, WA.

Gilbert FF, Nancekivell EG. 1982. Food habits of mink (*Mustela vison*) and otter (*Lutra canadensis*) in northeastern Alberta. Can J Zool 60:1282-1288.

Golet G, Kuletz K, Roby D, Irons D. 2000. Adult prey choice affects chick growth and reproductive success in pigeon guillemots. Auk 117:82-88.

Greer KR. 1955. Yearly food habits of the river otter in the Thompson Lakes region, Northwestern Montana, as indicated by scat analyses. Am Midl Nat 54:299-313.

Hart Crowser. 2005. Post-dredge sediment characterization, Integrated Support Command, Seattle Pier 36 Facility, US Coast Guard, Facilities Design and Construction Center, Seattle, Washington. Hart Crowser, Inc., Seattle, WA.

Hotchkiss D. 2007. Personal communication (reports from dock workers regarding the presence of pigeon guillemot nests under the pier at T-18). Project manager, Port of Seattle, Seattle, WA.

Jarvinen AW, Ankley GT. 1999. Linkage of effects to tissue residues: Development of a comprehensive database for aquatic organisms exposed to inorganic and organic chemicals. SEATAC Press, Pensacola, FL.

Jeffries SJ. 2001. Personal communication (telephone conversation with Berit Bergquist, Windward Environmental LLC, Seattle, WA). Research Scientist, Washington Department of Fish and Wildlife, Olympia, WA. October 29.

Jensen GC. 2008. Personal communication (telephone conversation with M. Luxon, Windward, regarding coonstripe shrimp ecology. School of Aquatic and Fishery Science, University of Washington, Seattle, WA. October 15, 2008.

Johnson L. 2006. Personal communication (telephone conversation with Matt Luxon, Windward Environmental LLC, regarding incidental sediment ingestion by English sole and Pacific staghorn sculpin). Zoologist, Ecotoxicology and Environmental Fish Health Program, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, Seattle, WA. May 17, 2006.

Kenney MJ. 1982. East, West and Duwamish Waterways navigation improvement study: fish and wildlife coordination act report. Division of Ecological Services, US Fish and Wildlife Service, Olympia, WA.

King County. 1996. Norfolk CSO sediment remediation project five-year monitoring program. 1996 sampling for bioassay and subsequent analysis for chemistry. Prepared for Elliott Bay/Duwamish Restoration Program. King County Department of Natural Resources, Seattle, WA.

King County. 1999. King County combined sewer overflow water quality assessment for the Duwamish River and Elliott Bay. Vol 1, Appendix B2, B3, & B4: human health, wildlife, and aquatic life risk assessments. King County Department of Natural Resources, Seattle, WA.

Kuletz K. 1998. Pigeon guillemot (*Cepphus columba*). Restoration Notebook. November 1998. Exxon Valdez Oil Spill Trustee Council.

Kurta A. 1995. Mammals of the Great Lakes region. University of Michigan Press, Ann Arbor, MI.

Lange J. 2006. Personal communication (telephone conversation with Matt Luxon, Windward Environmental LLC, regarding incidental sediment ingestion by

English sole and Pacific staghorn sculpin). Zoologist, Ecotoxicology and Environmental Fish Health Program, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, Seattle, WA. May 18, 2006.

Larsen DN. 1984. Feeding habits of river otters in coastal southeastern Alaska. J Wildl Manage 484:1460-1452.

Litzow M, Piatt J, Abookire A, Prichard A, Robards M. 2000. Monitoring temporal and spatial variability in sandeel (*Ammodytes hexapterus*) abundance with pigeon guillemot (*Cepphus columba*) diets. ICES J Mar Sci 57:976-986.

Litzow MA, Piatt JF. 2003. Variance in prey abundance influences time budgets of breeding seabirds: evidence from pigeon guillemots *Ceppus columba*. J Avian Biol 34:54-64.

Lowry LF, Frost KJ. 1981. Feeding and trophic relationships of phocid seals and walruses in the Eastern Bering Sea. In: Hood DW, Calder JA, eds, The Eastern Bering Sea shelf: oceanography and resources. Vol 2. Department of Commerce, Washington, DC, pp 813-824.

Matthews KR. 1990. An experimental study of the habitat preferences and movement patterns of copper, quillback, and brown rockfishes (*Sebastes* spp.). Environ Biol Fish 29:161- 78.

Meador JP. 2006. Rationale and procedures for using the tissue-residue approach for toxicity assessment and determination of tissue, water, and sediment quality guidelines for aquatic organisms. Human Ecol Risk Assess 12:1018-1073.

Melquist WE, Hornocker MG. 1983. Ecology of river otters in west central Idaho. In: Kirkpatrick RL, ed, Wildlife monographs. Vol 83. The Wildlife Society, Bethesda, MD, p 60.

METRO. 1995. CSO overflow monitoring program: CSO monitoring results (1986-1991). METRO, Seattle, WA.

Meyer JS, Adams WJ, Brix KV, Luoma SN, Mount DR, Stubblefield WA, Wood CM, eds. 2005. Toxicity of dietborne metals to aquatic organisms. Chapter 1, introduction, pp. 1-11. Society of Environmental Toxicology and Chemistry (SETAC), Pensacola, FL.

Michelsen TC, Bragdon-Cook K. 1993. Technical information memorandum: Organic carbon normalization of sediment data. Washington Department of Ecology, Olympia, WA.

Miller BS, Simenstad CA, Moulton LL, Fresh KL, Funk FC, Karp WA, Borton SF. 1977. Puget Sound baseline program nearshore fish survey. Final report, July 1974-June 1977. Prepared for Washington Department of Ecology. Fisheries Research Institute, University of Washington, Seattle, WA.

Mowbray EE, Pursley D, Chapman JA. 1979. The status, population characteristics, and harvest of river otters in Maryland. Maryland Wildlife Administration. Waverly Press, Bethesda, MD.

Nagy KA. 1987. Field metabolic rate and food requirement scaling in mammals and birds. Ecol Monogr 57(2):111-128.

Nebraska Game and Parks Commission. 2000. The river otter. [Cited June 6, 2000.] Available from: http://ngp.ngpc.state.ne.us/wildlife/otters.html.

NMFS. 1997. Impacts of California sea lions and Pacific harbor seals on salmonids and on the coastal ecosystems of Washington, Oregon, and California. NOAA Technical Memorandum NMFS-NWFSC-28. National Marine Fisheries Service, Seattle, WA.

Pitcher KW. 1980. Food of the harbor seal, *Phoca vitulina richardsi*, in the Gulf of Alaska. USNMFS Bull 78:544-549.

Pitcher KW, Calkins OG. 1979. Biology of the harbor seal, Phoca vitulina richardsi, in the Gulf of Alaska. Final. Outer Continental Shelf Environmental Assessment Program Research Unit 229. Bureau of Land Management, US Department of the Interior, Washington, DC.

Poole A. 1989. Ospreys: a natural and unnatural history. Cambridge University Press, Cambridge, UK.

Poole AF. 1983. Courtship feeding, clutch size, and egg size in ospreys: a preliminary report. In: Bird DM, Seymour NR, Gerrard JM, eds, Biology and management of bald eagles and ospreys. Harpell Press, Saint-Anne-de-Bellevue, Quebec, pp 243-256.

Poole AF, Bierregaard RO, Martell MS. 2002. Osprey (*Pandion haliaetus*). In: Poole A, ed, The birds of North America online. Cornell Laboratory of Ornithology, Ithaca, NY.

PSEP. 1995. Recommended guidelines for conducting laboratory bioassays on Puget Sound sediments. Final Report. Prepared for the Puget Sound Estuary Program, U.S. Environmental Protection Agency, Region 10, Office of Puget Sound, and U.S. Army Corps of Engineers, Seattle District, Seattle, WA. PTI Environmental Services, Inc., Seattle, WA.

Roffe TJ, Mate BR. 1984. Abundances and feeding habits of pinnipeds in the Rogue River, Oregon. J Wildl Manage 48:1262-1274.

Sample BE, Opresko DM, Suter GW. 1996. Toxicological benchmarks for wildlife. 1996 revision. ES/ERM-86/R3. Office of Environmental Management, US Department of Energy, Washington, DC.

Schaffer KE. 1989. Seasonal and size variations in diets of harbor seals. 8th Biennial Conference on the Biology of Marine Mammals, December 7-11, 1989, Pacific Groves, CA, p 62.

SEA. 2000. East Waterway channel deepening project, Stage 1. Data report: water quality monitoring during dredging at Terminal 18, East Waterway, Seattle, Washington. Prepared for US Army Corps of Engineers, Seattle District. Striplin Environmental Associates, Inc., Olympia, WA.

Spinola RM, Serfass TL, Brooks RP. 1999. Radiotelemetry study: river otters reintroduction at Letchworth State Park. Progress report no. 1. New York State Department of Environmental Conservation, New York River Otter Project, and New York State Office of Parks, Recreation, and Historic Preservation, Albany, NY.

Stenson GB, Badgero GA, Fisher HD. 1984. Food habits of the river otter *Lutra canadensis* in the marine environment of British Columbia. Can J Zool 62:88-91.

Stewart BS, Leatherwood SL, Yochem PK. 1989. Harbor seal tracking and telemetry by satellite. Mar Mam Sci 5:361-375.

Strand J. 1999. Personal observation. In: King County combined sewer overflow water quality assessment for the Duwamish River and Elliott Bay. Appendix B2, B3, & B4: Human Health, Wildlife, and aquatic life risk assessments. King County Department of Natural Resources, Seattle, WA.

Tabor JE, Wight HM. 1977. Population status of river otter in western Oregon. J Wildl Manage 41:692-699.

Tabor R, McCoy C, Camacho S, Celedonia M, Vecht S. 2004. Habitat use and movements of juvenile chinook salmon in south Lake Washington, 2003 investigations. Presentation at 2004 Greater Lake Washington Chinook Workshop, Shoreline Center, Shoreline, Washington, February 2, 2004 [online]. City of Seattle, Seattle, WA. Available from: http://www.cityofseattle.net/salmon/docs/workshop04/abstracts/Tabor2.pdf.

Tetra Tech. 1996. Preliminary sediment survey for Pier 36 modification, US Coast Guard Support Center, Seattle, Washington. Tetra Tech, Inc., Bellevue, WA.

Tetra Tech. 1997. Surface sediment investigation, Piers 36 and 37, US Coast Guard Integrated Support Center, Seattle, Washington. Tetra Tech, Inc., Bellevue, WA.

Thompson P. 2006. Personal communication (telephone conversation with Angelita Rodriquez, Windward Environmental, regarding current number of osprey nests on the Lower Duwamish Waterway). Urban Biologist, Washington Department of Fish and Wildlife, Mill Creek, WA. June 20, 2006.

Toweill DE. 1974. Winter food habits of river otters in western Oregon. J Wildl Manage 38(1):107-111.

Van Daele LJ, Van Daele HA. 1982. Factors affecting the productivity of ospreys nesting in west-central Idaho. Condor 84:292-299.

Van den Berg M, Birnbaum L, Bosveld ATC, Brunström B, Cook P, Feeley M, Giesy JP, Hanberg A, Hasegawa R, Kennedy S, Kubiak T, Larsen JC, van Leeuwen FXR, Djien Liem AK, Nolt C, Peterson RE, Poellinger L, Safe S, Schrenk D, Tillitt D, Tysklind M, Younes M, Waern F, Zacharewski T. 1998. Toxic equivalency factors (TEFs) for PCBs, PCDDs, PCDFs for humans and wildlife. Environ Health Perspect 106(12):775-792.

Van den Berg M, Birnbaum LS, Denison M, De Vito M, Farland W, Feeley M, Fiedler H, Hakansson H, Hanberg A, Haws L, Rose M, Safe S, Schrenk D, Tohyama C, Tritscher A, Tuomisto J, Tysklind M, Walker N, Peterson RE. 2006. The 2005 World Health Organization reevaluation of human and mammalian toxic equivalency factors for dioxins and dioxin-like compounds. Tox Sci 93(2):223-241.

Walker B. 1999. Monitoring of 1998-99 California sea lion predation in the Lake Washington estuary and the lower Duwamish River. Washington Department of Fish and Wildlife, Lacey, WA.

Windward. 2001. Terminal 18 deepening project. Post-dredge monitoring of Port of Seattle East Waterway deepening, stage 1 dredging area. Data report. Prepared for Port of Seattle. Windward Environmental LLC, Seattle, WA.

Windward. 2002a. East Waterway, Harbor Island Superfund site: Nature and extent of contamination. Surface sediment data report - phases 1 and 2. Prepared for the Port of Seattle. Windward Environmental LLC, Seattle, WA.

Windward. 2002b. East Waterway, Harbor Island Superfund site: Technical memorandum: Tissue chemistry results for juvenile chinook salmon collected from Kellogg Island and East Waterway. Prepared for the Port of Seattle. Windward Environmental LLC, Seattle, WA.

Windward. 2004. Lower Duwamish Waterway remedial investigation. Juvenile chinook salmon data report. Prepared for Lower Duwamish Waterway Group. Windward Environmental LLC, Seattle, WA.

Windward. 2005. Quality assurance project plan: East Waterway fish tissue collection and chemical analyses. Prepared for the Port of Seattle. Windward Environmental LLC, Seattle, WA.

Windward. 2006. East Waterway, Harbor Island Superfund site. Technical memorandum: tissue chemistry results for East Waterway fish tissue samples. Windward Environmental LLC, Seattle, WA.

Windward. 2007a. East Waterway Harbor Island Superfund site: Slip 27 sediment sampling results. Windward Environmental LLC, Seattle, WA.

Windward. 2007b. East Waterway Phase 1 removal action: recontamination monitoring 2006 data report. Windward Environmental LLC, Seattle, WA.

Windward. 2008a. East Waterway Phase 1 removal action: recontamination monitoring 2007 data report. Windward Environmental LLC, Seattle, WA.

Windward. 2008b. Quality assurance project plan: benthic invertebrate tissue/gastropod collection. East Waterway Operable Unit supplemental remedial investigation/feasibility study. Windward Environmental LLC, Seattle, WA.

Windward. 2008c. Quality assurance project plan: clam studies. East Waterway Operable Unit supplemental remedial investigation/feasibility study. Windward Environmental LLC, Seattle, WA.

Windward. 2008d. Quality assurance project plan: fish and shellfish tissue collection and chemical analysis. East Waterway Operable Unit supplemental remedial investigation/feasibility study. Windward Environmental LLC, Seattle, WA.

Windward. 2009. Quality assurance project plan: surface water collection and chemical analysis. Final. East Waterway Operable Unit supplemental remedial investigation/feasibility study. Windward Environmental LLC, Seattle, WA.

Wingert RC, Terry CB, Miller BS. 1979. Food and feeding habits of ecologically important nearshore and demersal fishes in central Puget Sound. FRI-UW-7903. Prepared for Washington Department of Ecology. Fisheries Research Institute, University of Washington, Seattle, WA.

Wise MH, Linn IJ, Kennedy CR. 1981. A comparison of the feeding biology of mink *Mustela vison* and otter *Lutra lutra*. J Zool London 195:181-213.